SECTION 6

HEALTH AND ENVIRONMENTAL EFFECTS

6.0 Health and Environmental Effects

6.1 Health Effects and Radiation Dosimetry

6.1.1 Radioactive Airborne Emissions

We used data on radioactive emissions (Section 3) to estimate the public health impact of these emissions. Our assessments include estimates of the following radiation exposures and health risks:

1. Dose equivalent rates and working level exposures to the most exposed individuals (maximum individual) and to the average exposed individuals in the regional population (average individual)
2. Collective dose equivalent rates and working level exposures to the regional population
3. Lifetime fatal cancer risks to the maximum and average individuals in the regional population
4. Genetic effect risk to the descendants of the maximum and average individuals in the regional population
5. The number of fatal cancers committed in the regional population per year of model mine operation
6. The number of genetic effects committed to the descendants of the regional population per year of model mine operation

The somatic health impact risks estimated in this report are for fatal cancers only. For whole body exposure, the risk of nonfatal cancer is about the same or slightly less than for fatal cancer. Thus, for whole body doses, it is conservatively estimated that one nonfatal cancer could occur for each additional fatal cancer. The somatic health impact for the regional population (additional cancers per year) is calculated at equilibrium for continuous exposure and this is equal to the additional cancers committed over all time per year of exposure; thus we used the term committed additional cancers (see Appendix L).

The genetic effect risks estimated in this report are for effects in descendants of an irradiated parent or parents. Genetic effects per year in the regional population due to radionuclide releases from the mines are calculated for an equilibrium exposure situation. The calculated genetic effects per year at equilibrium is equal to the genetic effects committed over all time from one year exposure. Thus, the calculated additional

Table 6.1 Annual release rates (Ci) used in the dose equivalent and health effects computations for active uranium mines

Classification	Location	Average Surface Mine[a]			Average Large Surface Mine[a]		
		U	Th	Rn-222	U	Th	Rn-222
Mining activities	Pit/mine site	4.3E-3	2.2E-4	1.99E+2	2.57E-2	1.44E-3	7.97E+2
Ore	Pile site	1.01E-2	1.42E-4	4.2E+1	4.42E-2	6.20E-4	9.6E+1
Sub-ore	Pile site	4.2E-4	8.4E-6	5.0E+1	1.51E-3	3.00E-5	1.66E+2
Overburden/ waste rock	Pile site	2.25E-3	1.50E-4	4.0E+1	1.34E-2	8.94E-4	2.02E+2
Vehicular dust	Mining area	9.9E-4	3.7E-4	0	5.86E-3	2.17E-3	0
Total	All sources	1.81E-2	8.90E-4	3.31E+2	9.07E-2	5.15E-3	1.26E+3

[a] Release rates taken from Tables 3.32 to 3.35.
[b] Release rates taken from Tables 3.51 and 3.54 to 3.56.

Table 6.1 (cont.)

Average Underground Mine [b]			Average Large Underground Mine [b]			In Situ Leach Mine [c]		
U	Th	Rn-222	U	Th	Rn-222	U	Th	Rn-222
2.22E-4	2.8E-6	3.08E+2	2.41E-3	3.10E-5	3.42E+3	1.0E-1	0	6.50E+2
9.63E-4	1.35E-5	7.7	1.07E-2	1.50E-4	6.83E+1	N.A. [d]	N.A.	N.A.
1.04E-3	8.4E-6	6.1E+1	5.95E-3	4.8E-5	3.38E+2	N.A.	N.A.	N.A.
9.6E-6	6.4E-7	5.0E-1	5.10E-5	3.40E-6	2.6	N.A.	N.A.	N.A.
6.5E-5	2.4E-5	0	1.29E-4	4.80E-5	0	N.A.	N.A.	N.A.
2.30E-3	4.93E-5	3.77E+2	1.92E-2	2.80E-4	3.83E+3	1.0E-1	0	6.50E+2

[c] Release rates taken from Table 3.59.

[d] N.A.- Not Applicable.

Note.--Columns labeled U and Th include each daughter of the decay chain in secular equilibrium.

Table 6.2 Annual release rates (Ci) used in the dose equivalent and health effects computations for inactive uranium mines

	Surface Mine [a]			Underground Mine [b]		
Location	U	Th	Rn-222	U	Th	Rn-222
Pit/vents-portals	0	0	8.1	0	0	7.55
Waste rock/sub-ore pile	1.48E-3	1.1E-5	1.74E+1	2.38E-4	1.7E-6	1.7

(a) Release rates taken from Tables 3.70 and 3.74.

(b) Release rates taken from Tables 3.76 and 3.77.

Note.--Column headings U and Th include each daughter of the decay chain in secular equilibrium.

genetic effects are committed effects to all future generations for one year of exposure to the regional population.

We calculated individually each major source of radionuclide airborne emissions for each model uranium mine site so that we could determine the extent that each source contributed to the total health impact. Tables 6.1 and 6.2 contain the annual release rates for each source classification (or location) that we used to calculate dose equivalent rates and health effects for active and inactive uranium mines.

The estimated annual working level exposures from Rn-222 emissions by the model uranium mines are listed in Table 6.3. The working level exposures presented for the maximum individual are the Rn-222 decay product levels to which an individual would be continuously exposed for an entire year. Working level exposure to the regional population is the sum of the exposures to all individuals in the exposed population from the annual release from the model mine.

We estimated radiological impacts of radioactive airborne emissions from the model uranium mines with the AIRDOS-EPA (Mo79), RADRISK (Du80), and DARTAB (Be80) computer codes. Appendixes K and L contain explanations of our use of these computer codes.

Where emissions for U-238 plus daughters and Th-232 plus daughters were reported (Section 3), a source term for both the parent and important daughters were input into the AIRDOS-EPA code. For example, a reported emission rate of 0.01 Ci/yr of U-238 plus daughters (U in Tables 6.1 and 6.2) would be input into the AIRDOS-EPA code as 0.01 Ci/yr of U-238, 0.01 Ci/yr of U-234, 0.01 Ci/yr of Th-230, 0.01 Ci/yr of Ra-226, 0.01 Ci/yr of Pb-214, 0.01 Ci/yr of Bi-214, 0.01 Ci/yr of Pb-210, and 0.01 Ci/yr of Po-210. A reported emission rate of 0.01 Ci/yr of Th-232 plus daughters (Th in Tables 6.1 and 6.2) would be input into the AIRDOS-EPA code as 0.01 Ci/yr of Th-232, 0.01 Ci/yr of Ra-228, 0.01 Ci/yr of Ac-228, 0.01 Ci/yr of Th-228, 0.01 Ci/yr of Ra-224, 0.01 Ci/yr of Pb-212, 0.01 Ci/yr of Bi-212, and 0.0036 Ci/yr of Tl-208. The Tl-208 source term is approximately one-third that of Bi-212 because of the branching ratio.

The maximum individual, average individual, and population dose equiv-

Table 6.3 Annual working level exposure from radon-222 emissions from model uranium mines

Source	Maximum Individual (WL)[a]	Average Individual (WL)	Regional Population (person-WL)
Average Surface Mine	2.3E-4	4.5E-7	6.5E-3
Average Large Surface Mine	8.4E-4	1.7E-6	2.5E-2
Average Underground Mine	4.6E-4	2.1E-6	7.5E-2
Average Large Underground Mine	4.7E-3	2.1E-5	7.6E-1
Inactive Surface Mine	1.8E-5	3.5E-8	5.0E-4
Inactive Underground Mine	1.1E-5	5.1E-8	1.8E-3
In Situ Leach Mine	4.5E-4	8.9E-7	1.3E-2

[a] Working level.

Table 6.4 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average surface uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	2.4	5.4E-3	7.7E-2
Endosteal	3.4E+1	7.5E-2	1.1
Pulmonary	1.2E+1	6.3E-3	9.0E-2
Muscle	5.5E-1	2.0E-3	2.7E-2
Liver	1.6	6.3E-3	9.1E-2
Stomach wall	9.7E-2	8.9E-5	1.3E-3
Pancreas	5.2E-1	1.9E-3	2.7E-2
LLI[(a)] wall	4.6E-1	1.6E-3	2.3E-2
Kidney	4.2	1.8E-2	2.5E-1
Bladder wall	3.0E-1	9.7E-4	1.4E-2
ULI[(b)] wall	2.1E-1	5.2E-4	7.4E-3
SI[(c)] wall	9.4E-2	1.2E-4	1.7E-3
Ovaries	5.1E-1	1.9E-3	2.7E-2
Testes	5.4E-1	1.9E-3	2.7E-2
Spleen	6.4	2.8E-2	4.0E-1
Uterus	5.1E-1	1.9E-3	2.7E-2
Thymus	5.2E-1	1.9E-3	2.7E-2
Thyroid	5.4E-1	1.9E-3	2.7E-2
Weighted mean	4.9	5.5E-3	7.8E-2

(a) Lower large intestine wall.
(b) Upper large intestine wall.
(c) Small intestine wall.

Table 6.5 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average large surface uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	1.35E+1	2.7E-2	3.9E-1
Endosteal	1.9E+2	3.8E-1	5.4
Pulmonary	6.6E+1	3.1E-2	4.5E-1
Muscle	3.0	9.6E-3	1.4E-1
Liver	8.9	3.2E-2	4.6E-1
Stomach wall	5.4E-1	4.5E-4	6.4E-3
Pancreas	3.0	9.6E-3	1.4E-1
LLI wall	2.5	8.2E-3	1.2E-1
Kidney	2.1E+1	9.0E-2	1.3
Bladder wall	1.7	4.9E-3	7.0E-2
ULI wall	1.1	2.6E-3	3.8E-2
SI wall	5.2E-1	6.0E-4	8.6E-3
Ovaries	2.8	9.6E-3	1.4E-1
Testes	3.0	9.6E-3	1.4E-1
Spleen	3.5E+1	1.4E-1	2.0
Uterus	2.8	9.6E-3	1.4E-1
Thymus	2.9	9.6E-3	1.4E-1
Thyroid	3.0	9.6E-3	1.4E-1
Weighted mean	2.7E+1	2.7E-2	3.8E-1

Table 6.6 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average underground uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	5.1E-1	8.3E-4	2.9E-2
Endosteal	7.2	1.2E-2	4.1E-1
Pulmonary	2.9	5.0E-3	1.8E-1
Muscle	1.2E-1	2.3E-4	8.3E-3
Liver	3.5E-1	7.2E-4	2.7E-2
Stomach wall	2.0E-2	2.8E-5	1.0E-3
Pancreas	1.1E-1	2.2E-4	8.0E-3
LLI wall	9.4E-2	1.8E-4	6.5E-3
Kidney	9.1E-1	2.0E-3	7.4E-2
Bladder wall	6.4E-2	1.2E-4	4.4E-3
ULI wall	4.3E-2	7.3E-5	2.7E-3
SI wall	2.0E-2	2.8E-5	1.0E-3
Ovaries	1.1E-1	2.2E-4	8.0E-3
Testes	1.1E-1	2.3E-4	8.0E-3
Spleen	1.4	3.1E-3	1.1E-1
Uterus	1.1E-1	2.2E-4	7.9E-3
Thymus	1.1E-1	2.2E-4	8.0E-3
Thyroid	1.1E-1	2.3E-4	8.1E-3
Weighted mean	1.1	2.0E-3	7.1E-2

Table 6.7 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average large underground uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	4.2	6.9E-3	2.5E-1
Endosteal	6.0E+1	9.6E-2	3.5
Pulmonary	2.5E+1	4.7E-2	1.7
Muscle	9.7E-1	1.9E-3	6.9E-2
Liver	2.9	6.0E-3	2.2E-1
Stomach wall	1.7E-1	2.3E-4	8.5E-3
Pancreas	9.4E-1	1.8E-3	6.8E-2
LLI wall	7.8E-1	1.5E-3	5.5E-2
Kidney	7.7	1.7E-2	6.2E-1
Bladder wall	5.4E-1	1.0E-3	3.6E-2
ULI wall	3.6E-1	6.0E-4	2.2E-2
SI wall	1.6E-1	2.3E-4	8.4E-3
Ovaries	9.2E-1	1.8E-3	6.6E-2
Testes	9.4E-1	1.8E-3	6.8E-2
Spleen	1.2E+1	2.6E-2	9.2E-1
Uterus	9.2E-1	1.8E-3	6.6E-2
Thymus	9.4E-1	1.8E-3	6.7E-2
Thyroid	9.4E-1	1.9E-3	6.8E-2
Weighted mean	9.8	1.8E-2	6.2E-1

Table 6.8 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model inactive surface uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	2.1E-1	4.8E-4	6.9E-3
Endosteal	2.9	6.8E-3	9.8E-2
Pulmonary	9.5E-1	5.0E-4	7.2E-3
Muscle	5.6E-2	1.8E-4	2.6E-3
Liver	1.4E-1	5.5E-4	7.8E-3
Stomach wall	1.5E-2	1.1E-5	1.6E-4
Pancreas	5.4E-2	1.8E-4	2.6E-3
LLI wall	4.4E-2	1.4E-4	2.0E-3
Kidney	3.5E-1	1.5E-3	2.1E-2
Bladder wall	3.3E-2	9.2E-5	1.3E-3
ULI wall	2.4E-2	4.7E-5	6.7E-4
SI wall	1.4E-2	1.3E-5	1.8E-4
Ovaries	5.2E-2	1.8E-4	2.5E-3
Testes	5.5E-2	1.8E-4	2.6E-3
Spleen	5.3E-1	2.3E-3	3.3E-2
Uterus	5.2E-2	1.8E-4	2.5E-3
Thymus	5.3E-2	1.8E-4	2.5E-3
Thyroid	5.5E-2	1.8E-4	2.6E-3
Weighted mean	3.9E-1	4.7E-4	6.8E-3

Table 6.9 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model inactive underground uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	5.8E-2	9.3E-5	3.4E-3
Endosteal	8.0E-1	1.3E-3	4.6E-2
Pulmonary	2.7E-1	3.4E-4	1.3E-2
Muscle	1.6E-2	2.9E-5	1.0E-3
Liver	3.9E-2	7.9E-5	2.8E-3
Stomach wall	4.0E-3	5.2E-6	1.8E-4
Pancreas	1.5E-2	2.8E-5	1.0E-3
LLI wall	1.2E-2	2.2E-5	8.0E-4
Kidney	9.7E-2	2.1E-4	7.6E-3
Bladder wall	9.1E-3	1.6E-5	5.8E-4
ULI wall	6.6E-3	1.0E-5	3.7E-4
SI wall	3.7E-3	4.9E-6	1.8E-4
Ovaries	1.4E-2	2.7E-5	9.7E-4
Testes	1.5E-2	2.8E-5	1.0E-3
Spleen	1.5E-1	3.2E-4	1.2E-2
Uterus	1.4E-2	2.7E-5	9.8E-4
Thymus	1.5E-2	2.8E-5	1.0E-3
Thyroid	1.5E-2	2.8E-5	1.0E-3
Weighted mean	1.1E-1	1.5E-4	5.7E-3

Table 6.10 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a hypothetical in situ uranium solution mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	1.6E-1	2.7E-4	3.8E-3
Endosteal	2.8	5.0E-3	7.1E-2
Pulmonary	3.9E+1	2.0E-2	2.9E-1
Muscle	8.4E-3	2.2E-5	3.1E-4
Liver	1.9E-2	5.4E-5	7.7E-4
Stomach wall	1.6E-2	5.7E-5	8.1E-4
Pancreas	7.6E-3	2.1E-5	3.0E-4
LLI wall	6.1E-1	2.5E-3	3.5E-2
Kidney	3.3E-1	1.0E-3	1.5E-2
Bladder wall	4.8E-3	1.2E-5	1.6E-4
ULI wall	2.0E-1	8.1E-4	1.2E-2
SI wall	3.6E-2	1.4E-4	2.0E-3
Ovaries	7.3E-3	2.1E-5	3.0E-4
Testes	8.9E-3	2.2E-5	3.1E-4
Spleen	4.6E-2	1.8E-4	2.5E-3
Uterus	7.4E-3	2.1E-5	3.0E-4
Thymus	7.9E-3	2.1E-5	3.0E-4
Thyroid	8.4E-3	2.1E-5	3.1E-4
Weighted mean	1.2E+1	6.2E-3	8.8E-2

alent rates* due to atmospheric radioactive particulate and Rn-222 emissions from the model uranium mine sites are presented in Tables 6.4 through 6.10. The Rn-222 dose equivalent rate is only for the inhalation and air immersion pathways and excludes Rn-222 daughters. The impact from Rn-222 daughters is addressed separately with a working level calculation. The dose equivalent estimates are for the model sites described for use with the AIRDOS-EPA code in Appendix K. Assumptions about food production and consumption for the maximum individual were selected for a rural setting. The maximum individual dose equivalent rate occurred about 1600 meters downwind from the center of the model site. The term "population" refers to the population living within a radius of 80 kilometers of the source. Population dose equivalents are the sum of the exposures to all individuals in the exposed population for the annual release from the model uranium mine.

Dose equivalent rates in Tables 6.4 through 6.10 indicate that the red marrow, endosteal cells, lung, kidneys, and spleen are generally the highest exposed target organs. A dose equivalent rate is presented for the "weighted mean" target organ, but this calculated result was not used in the health effect calculations. We calculated "weighted mean" dose equivalents by using organ dose equivalent weighting factors (see Appendix L) and summing the results. The weighted mean dose equivalent rate was presented instead of the total body dose equivalent rate.

Individual lifetime fatal cancer risks and estimated additional fatal cancers to the regional population due to atmospheric radioactive emissions from the model uranium mine sites are presented in Tables 6.11 and 6.12. The individual lifetime risks in Table 6.11 are those that would result from one year of exposure (external and internal) and the working levels estimated for those individuals. Except for the in situ leach mine, the individual lifetime risks in Table 6.12 are those that would result from a lifetime of exposure (71 years average life expectancy). The individual lifetime risks in Table 6.12 for the in situ leach mine are based on an exposure time of 18 years, which is the expected life, including restoration, of this type of model uranium mine.

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*The dose equivalent rates were not used to calculate risk and are only presented for perspective purposes. Risks of health impact were calculated directly from external and internal radionuclide exposure data.

Table 6.11 Individual lifetime fatal cancer risk for one year of exposure and estimated additional fatal cancers to the regional population due to annual radioactive airborne emissions from model uranium mines

Source	Maximum Exposed Individual	Average Exposed Individual	Regional Population
Average surface mine			
Particulates and Rn-222	6.7E-7	7.5E-10	1.1E-5
Radon-222 daughters	5.5E-6	1.1E-8	1.6E-4
Total	6.2E-6	1.2E-8	1.7E-4
Average large surface mine			
Particulates and Rn-222	3.7E-6	3.7E-9	5.4E-5
Radon-222 daughters	1.9E-5	4.1E-8	5.9E-4
Total	2.3E-5	4.5E-8	6.4E-4
Average underground mine			
Particulates and Rn-222	1.6E-7	2.8E-10	1.0E-5
Radon-222 daughters	1.1E-5	4.9E-8	1.7E-3
Total	1.1E-5	4.9E-8	1.7E-3
Average large underground mine			
Particulates and Rn-222	1.4E-6	2.5E-9	9.0E-5
Radon-222 daughters	1.1E-4	5.0E-7	1.8E-2
Total	1.1E-4	5.0E-7	1.8E-2
Inactive surface mine			
Particulates and Rn-222	5.5E-8	6.4E-11	9.1E-7
Radon-222 daughters	4.2E-7	8.3E-10	1.2E-5
Total	4.7E-7	8.9E-10	1.3E-5
Inactive underground mine			
Particulates and Rn-222	1.5E-8	2.0E-11	7.4E-7
Radon-222 daughters	2.7E-7	1.2E-9	4.4E-5
Total	2.8E-7	1.2E-9	4.5E-5
In situ leaching facility			
Particulates and Rn-222	1.6E-6	8.7E-10	1.2E-5
Radon-222 daughters	1.1E-5	2.1E-8	3.0E-4
Total	1.3E-5	2.2E-8	3.1E-4

Table 6.12 Individual lifetime fatal cancer risk due to lifetime exposure to radioactive airborne emissions from model uranium mines

Source	Maximum Exposed Individual	Average Exposed Individual (c)
Average surface mine(a)		
Particulates and Rn-222	1.4E-5	1.6E-8
Radon-222 daughters	1.2E-4	2.3E-7
Total	1.3E-4	2.5E-7
Average large surface mine(a)		
Particulates and Rn-222	6.6E-5	6.6E-8
Radon-222 daughters	3.5E-4	7.4E-7
Total	4.2E-4	8.1E-7
Average underground mine(a)		
Particulates and Rn-222	3.5E-6	5.8E-9
Radon-222 daughters	2.0E-4	9.0E-7
Total	2.0E-4	9.1E-7
Average large underground mine(a)		
Particulates and Rn-222	2.5E-5	4.4E-8
Radon-222 daughters	1.9E-3	8.6E-6
Total	1.9E-3	8.6E-6
Inactive surface mine(b)		
Particulates and Rn-222	3.9E-6	4.5E-9
Radon-222 daughters	3.0E-5	5.9E-8
Total	3.4E-5	6.3E-8
Inactive underground mine(b)		
Particulates and Rn-222	1.1E-6	1.4E-9
Radon-222 daughters	1.9E-5	8.5E-8
Total	2.0E-5	8.6E-8
In situ leaching facility(d)		
Particulates and Rn-222	1.6E-5	8.7E-9
Radon-222 daughters	2.0E-4	3.8E-7
Total	2.2E-4	3.9E-7

(a) Considers exposure for 17 years to active mining and 54 years to inactive mine effluents.

(b) Considers exposure for 71 years to inactive mine effluents.

(c) Considers the average individual in the regional population within an 80-km radius of the model mine.

(d) Considers 10-year operation and 8-year restoration.

Table 6.13 Genetic effect risk to descendants for one year of parental exposure to atmospheric radioactive airborne emissions from model uranium mines

Source	Descendants of Maximum Exposed Individual (effects/ birth)	Descendants of Average Exposed Individual (effects/ birth)	Descendants of Regional Population (effects/yr)
Average surface mine	6.3E-7	2.6E-9	1.6E-5
Average large surface mine	3.7E-6	1.3E-8	7.9E-5
Average underground mine	1.4E-7	2.9E-10	4.4E-6
Average large underground mine	1.1E-6	2.4E-9	3.6E-5
Inactive surface mine	6.0E-8	2.4E-10	1.4E-6
Inactive underground mine	1.6E-8	3.4E-11	5.0E-7
In situ leach facility	8.0E-9	2.7E-11	1.6E-7

Table 6.14 Genetic effect risk to descendants for a 30-year parental exposure to atmospheric radioactive airborne emissions from model uranium mines

Source	Effects/birth	
	Descendants of Maximum Exposed Individual	Descendants of Average Exposed Individual[c]
Average surface mine[a]	1.2E-5	4.6E-8
Average large surface mine[a]	6.4E-5	2.2E-7
Average underground mine[a]	2.6E-6	5.4E-9
Average large underground mine[a]	2.0E-5	4.0E-8
Inactive surface mine[b]	1.8E-6	7.2E-9
Inactive underground mine[b]	5.0E-7	5.8E-10
In situ leach facility[d]	1.4E-7	4.8E-10

[a]Considers exposure to 17 years active mining and 13 years inactive mine effluents.

[b]Considers exposure for 30 years to inactive mine effluents.

[c]Considers the average individual in the regional population within an 80-km radius of the model mine.

[d]Considers 10-year operation and 8-year restoration.

Genetic effect risks due to atmospheric radioactive emissions from the model uranium mine sites are presented in Tables 6.13 and 6.14. The risks to descendants in Table 6.13 are those that would result from one year of exposure to the parent or parents of first generation individuals. The descendant risks in Table 6.14 are those that would result from 30 years exposure to the first generation parent or parents, except for the in situ leach mine where we used an 18-year exposure time. The 30-year time period represents the mean years of life where gonadal doses are genetically significant.

We estimated the health impact risks with the DARTAB code using exposure data from the AIRDOS-EPA code. The dose equivalent and risk conversion factors that we used with the DARTAB code are tabulated in Appendix L. The somatic risk conversion factors are based on a lifetime (71 years average lifetime) exposure time, and the genetic effect risk conversion factors are based on a 30-year exposure time. When the exposure time for calculated risks was only one year, we calculated the risk by multiplying the risk calculated by DARTAB with the ratio of the one year exposure time to the exposure times used to calculate the risk conversion factors (1/71 for somatic effects and 1/30 for genetic effects to descendants of maximum and average exposed individuals).* Appendix L contains a discussion of the health risk assessment methodology.

We developed several tables to present the calculated health impact risk. The percentage contributions to the fatal cancer risks for individual sources at each model uranium mine site are contained in Table 6.15 for the maximum individual and Table 6.16 for the average individual. The fatal cancer risks by source term for one year of exposure which we used to calculate percentage contributions are contained in Tables L.4 to L.6 in Appendix L. Tables L.7 to L.9 contain genetic risks by source term at each model uranium mine site. The percent of the fatal cancer risk due to radon-222 daughter concentrations at model uranium mine sites is indicated in Table 6.17. The percent of the fatal cancer risk for principal nuclides and pathways due to radioactive particulate and Rn-222 emissions at each model uranium mine site are contained in Table 6.18.

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*A correction factor was not needed for DARTAB calculated genetic effects committed per year to the regional population.

Table 6.15 Percent of the fatal cancer risk for the maximum individual due to the sources of radioactive emissions at model uranium mines

Mine type	Percent of fatal cancer risk [a,b]				
	Mining Activities	Ore	Sub-ore	Spoils	Vehicular Dust
Average surface mine	56 (95)	18 (66)	14 (98)	12 (89)	<1 (0)
Average large surface mine	59 (93)	14 (41)	12 (98)	14 (86)	1 (0)
Average underground mine	80 (≧100)	3 (79)	17 (97)	<1 (96)	<1 (0)
Average large underground mine	89 (≧100)	2 (76)	9 (96)	<1 (96)	<1 (0)
Inactive surface mine	28[c] (100)	0	0	72 (84)	0
Inactive underground mine	77[c] (100)	0	0	23 (77)	0
In situ leach facility	100 (87)	0	0	0	0

[a] See Table L. 4, Appendix L.

[b] Values in parentheses are percent contribution of radon-222 daughters.

[c] Emissions from abandoned pit (surface mine) or vents and portals (underground mine).

Table 6.16 Percent of the fatal cancer risk for the average individual in the regional population due to the sources of radioactive emissions at model uranium mines

Mine type	Percent of fatal cancer risk[a,b]				
	Mining Activities	Ore	Sub-ore	Spoils	Vehicular Dust
Average surface mine	58 (97)	16 (78)	14 (99)	12 (93)	<1 (0)
Average large surface mine	60 (96)	11 (64)	12 (99)	16 (92)	1 (0)
Average underground mine	81 (≅100)	2 (93)	16 (99)	<1 (99)	<1 (0)
Average large underground mine	89 (≅100)	2 (91)	9 (99)	<1 (99)	<1 (0)
Inactive surface mine	29[c] (100)	0	0	71 (90)	0
Inactive underground mine	80[c] (100)	0	0	20 (92)	0
In situ leach facility	100 (96)	0	0	0	0

[a] See Table L.5, Appendix L.

[b] Values in parentheses are percent contribution of radon-222 daughters.

[c] Emissions from abandoned pit (surface mine) or vents and portals (underground mines).

Table 6.17 Percent of fatal cancer risks due to radon-222 daughter concentrations at model uranium mine sites

Source	Percent fatal cancer risk[a]
Average surface mine	89
Average large surface mine	84
Average underground mine	99
Average large underground mine	99
Inactive surface mine	88
Inactive underground mine	95
In situ leach facility	87

[a] Remainder due to radioactive particulate and Rn-222 emissions.

Table 6.18 Percent of the fatal cancer risk for principal nuclides and pathways due to radioactive particulate and Rn-222 emissions at model uranium mines

Mine Type	Receptor	Principal Nuclides	Percent of fatal cancer risk			
			Internal Pathways		External Pathways	
			Ingestion	Inhalation	Air Immersion	Ground Surface
Average Surface Mine	Max. Individual	U-238(20.0), U-234(22.1), Th-230(31.7), Ra-226(7.94), Po-210(7.33)	15.8	80.2	0.003	4.02
	Av. Individual or population	U-238(9.17), U-234(10.1), Th-230(22.7), Ra-226(21.3), Pb-210(6.92), Po-210(22.4)	60.1	38.1	0.005	1.81
Average Large Surface Mine	Max. Individual	U-238(20.0), U-234(22.2), Th-230(31.8), Ra-226(7.98)	15.9	80.0	0.002	4.05
	Av. Individual or Population	U-238(9.19), U-234(10.1), Th-230(22.7), Ra-226(21.4), Pb-210(6.94), Po-210(22.4)	60.5	37.7	0.004	1.83
Average Underground Mine	Max. Individual	U-238(17.9), U-234(19.8), Th-230(28.4), Ra-226(7.14), Po-210(6.59), Rn-222(13.6)	14.0	82.5	0.025	3.52
	Av. Individual or Population	U-238(12.0), U-234(13.2), Th-230(20.1), Ra-226(7.24), Po-210(7.31), Rn-222(34.6)	16.9	80.6	0.063	2.43
Average Large Underground Mine	Max. Individual	U-238(17.5), U-234(19.3), Th-230(27.7), Ra-226(6.97), Po-210(6.43), Rn-222(16.0)	13.6	82.9	0.029	3.39
	Av. Individual or Population	U-238(11.2), U-234(12.4), Th-230(18.8), Ra-226(6.76), Po-210(6.83), Rn-222(39.2)	15.7	82.0	0.071	2.25
Inactive Surface Mine[a]	Max. Individual	U-238(19.5), U-234(21.6), Th-230(31.0), Ra-226(9.4), Bi-214(5.31), Po-210(7.17)	16.8	75.4	0.002	7.85
	Av. Individual or Population	U-238(8.76), U-234(9.68), Th-230(21.7) Ra-226(26.1), Pb-210(6.77), Po-210(21.4)	62.2	34.3	0.003	3.48
Inactive Underground Mine[a]	Max. Individual	U-238(19.6), U-234(21.6), Th-230(31.1), Ra-226(9.45), Bi-214(5.33), Po-210(7.21)	16.9	75.2	0.001	7.88
	Av. Individual or Population	U-238(17.0), U-234(18.8), Th-230(28.5), Ra-226(12.7), Bi-214(4.81), Po-210(10.4)	26.3	66.6	0.004	7.09
In situ Leaching Facility	Max. Individual	U-238(45.2), U-234(50.0), U-235(2.21)	0.46	99.5	0.002	0.039
	Av. Individual or Population	U-238(43.3), U-234(47.8), U-235(2.12)	3.50	96.5	0.009	0.038

[a]Spoils source term only.

The fatal cancer health risk at each of the model uranium mine sites is dominated by the lung cancer risk from radon-222 daughter exposures (see Table 6.17). Radioactive particulates and Rn-222 contributed to a little over 10 percent of the total fatal cancer health risk at the model surface mines and at the in situ leaching facility (see Table 6.11). Essentially all the risks from the model underground mines are due to radon-222 daughter exposures. The fatal cancer health risks from the active model underground mines are greater than the risks from the active model surface mines because of the larger quantity of Rn-222 released. The risks are similar at inactive surface and underground mines.

The largest fatal cancer risk is from the average large underground mine (see Tables 6.11 and 6.12)--an estimated 1.9E-3 lifetime fatal cancer risk to the maximum exposed individual for a lifetime exposure. The lifetime fatal cancer risk to the average individual in the regional population is estimated to be 8.6E-6 for a lifetime exposure period. The number of estimated additional fatal cancers in the regional population per year of mine operation is estimated to be 1.8E-2.

For the active surface mines, about 60 percent of the radon daughter impact is from the exposed pit surfaces (see Table L.4). For the active underground mines, the predominate radon daughter impact is from mine vent air. For the inactive surface mine, about 70 percent of the radon daughter impact is from waste rock pile exhalation and about 30 percent was from the pit interior surfaces. About 80 percent of the radon daughter impact for the inactive underground mine was due to radon releases from the mine vents and entrance. The release of radon from the pregnant leach surge tanks was the predominate source of the radon daughter health impact risk for the model in situ leach mine. Detailed percentages of the lifetime fatal cancer risks by source term for each model uranium mine are contained in Tables 6.15 and 6.16.

The health impact from particulate radionuclides and Rn-222 was predominately due to U-238 and daughter radionuclides (see Table 6.18). Thorium-232 and daughters were only minor contributors to the particulate and Rn-222 fatal cancer risk with Rn-222 only contributing significantly (14 to 40 percent) at active underground mines. The majority of the exposure to individuals around the model uranium mines is received from the internal pathways. Inhalation was the most important internal pathway except for the average individual and regional population impact at surface mines

where ingestion was the major pathway (see Table 6.18). For active surface mines, about 52 percent of the particulate and Rn-222 impact to the maximum individual was from the ore source term, and about 25 percent of the health impact was from the mining activities source term (see Table L.4). For active underground mines, between 28 and 46 percent of the particulate and Rn-222 impact was from the ore source term and between 26 and 41 percent of the particulate and Rn-222 impact was from the sub-ore source term. The predominant source of the particulate and Rn-222 impact from the inactive mines was particulate radionuclides in wind-suspended dust from the waste rock pile. The release of particulate radionuclides from the uranium recovery plant was the predominant source of the particulate health impact risk for the model in situ leach mine.

For perspective, the calculated fatal cancer risks can be compared to the estimated cancer risk from all causes. The American Cancer Society estimates the risk of cancer death from all causes to be 0.15 (Ba79). The maximum exposed individual around the model average large underground mine is estimated to incur an additional lifetime fatal cancer risk of 0.0019 (1.3 percent) due to radioactive airborne emissions from the model mine. There is a regional population of 36,004 persons for the model average large underground mine site located in New Mexico. The cancer death rate for the State of New Mexico for whites of both sexes was 154.5 deaths per year for 1973 to 1976 per 100,000 people (NCI78). Applying this statistic to the regional population, about 56 cancer deaths are estimated to occur each year in the regional population from all causes. Applying the approximate fatal cancer risk coefficient of 0.15 to the regional population of 36,004 persons, about 5,400 people in the regional area would normally die of cancer. About 0.018 additional cancer deaths (0.00033 percent) in the regional population are estimated per year of operation from radioactive airborne emissions at the model average large underground mine.

The risk of genetic effects from radiation exposure at model uranium mine sites is very small compared to the normal occurrence of hereditary disease. The national incidence of genetic effects is 60,000 per 10^6 births (NAS72). The normal occurrence of hereditary disease for the descendants of the regional population of 14,297 at the model average large surface mine in Wyoming is 0.06 effects per birth and 12.1 effects per year, based on 202 live births per year in the regional population. (We present statistics for the site of the average large surface mine since the largest

genetic risk for all the evaluated model uranium mines occurred at this site [see Tables 6.13 and 6.14]). We estimated the genetic effect risk to the descendants of the maximum exposed individual to be an additional 6.4E-5 effects/birth (0.1 percent increase) for a 30-year exposure period. The genetic effect risk to the descendants of the average exposed individual in the regional population is estimated to be an additional 2.2E-7 effects/birth (0.00036 percent increase) for a 30-year exposure period. The number of additional genetic effects committed to the descendants of the regional population per year of operation of the average large surface mine is estimated to be 7.9E-5. The additional committed genetic effects constitute a very small increase to the 12.1 effects that will normally occur each year in the live births within the regional population.

6.1.2 Nonradioactive Airborne Emissions

To calculate atmospheric concentrations at the location of the maximum individual, we used the data on nonradioactive air pollutant emissions from Section 3. We compared these pollutant air concentrations with calculated nonoccupational threshold limit values, natural background concentrations, and average urban concentrations of selected airborne pollutants in the United States.

The "natural" background atmospheric concentration has been defined (Va71) as the concentration of pollutants in areas absent of activities by man which cause significant pollution. Variations in background levels may result from differences in mineral content of the soil, vegetation, wind conditions, and the proximity to the ocean or metropolitan areas. Based on an extensive literature survey and consideration of the abundance and distribution of the chemical elements in the ocean and earth's crust, a set of "natural" background airborne concentrations has been developed for the United States (Va71). Natural background airborne concentrations for selected pollutants are listed in the second column of Table 6.19. Also listed in the table are average concentrations of airborne pollutants in urban areas. The latter are arithmetic mean concentrations obtained from measurements taken over a period of several years (Va71).

6.1.2.1 Combustion Products

Airborne concentrations of combustion products released from diesel and gasoline-powered equipment were estimated for the site of the maximum

Table 6.19 Natural background concentrations and average urban concentrations of selected airborne pollutants in the United States

Pollutant	Natural Background Concentration, $\mu g/m^3$	Average Urban Concentration, $\mu g/m^3$
Gases		
CO	100	7000
NO_x	40	141
NH_3	10	80
SO_x	5	62
CO_2	594,000	NR
Hydrocarbons	NR(a)	500
Suspended particles		
Total	20 - 40	105
As	0.005	0.02 (1)
Ba	0.005	NR
Cd	0.0001	0.002
Co	0.0001	0.0005
Cr	0.001	0.015
Cu	0.01	0.09
Hg	0.0005	0.1
Fe	0.2 - 0.5	1.58
Pb	0.001	0.79
Mg	0.1	NR
Mn	0.01	0.1
Mo	0.0005	0.005
Ni	0.001	0.034
Se	0.001	NR
Sr	0.005	NR
Th	0.0005	NR
U	0.0001	NR
V	0.001	0.05
Zn	0.01	0.67
Zr	0.001	NR

(a) NR - Not Reported.
Source: Va71; except for CO_2, Ba76.

individual. The concentrations were computed using the annual release rates given in Tables 3.30 and 3.52 with dispersion parameters applicable for the model underground (New Mexico) and surface (Wyoming) mining areas (Appendix K). The estimated combustion product concentrations are low compared to the natural background and average urban concentrations (see Table 6.20). A conservative threshold limit value (TLV) was computed, as described in Section 6.1.2.3 for SO_2, CO, and NO_2. Of these pollutants, only the nitrogen oxide concentrations at the average large surface mine exceed the nonoccupational TLV. Considering these comparisons and the conservative nature of the analyses, combustion products released from heavy uranium mining equipment do not appear to pose a health hazard.

6.1.2.2 Nonradioactive Gases

Airborne concentrations of the three principal nonradioactive gases released from the hypothetical in situ leach mining site were computed using the source terms from Table 3.59 and the meteorological parameters and dispersion model described in Appendix K. Table 6.21 shows the estimated atmospheric concentrations at the location of a maximum individual; occupational threshold limit values (TLV's); adjusted TLV's applicable to nonoccupational exposures; and the percent the estimated concentrations are of the adjusted TLV's. The occupational TLV's have been conservatively adjusted. They were adjusted on the basis of a 168-hr week, instead of a 40-hour week and a safety factor of 100.

The results of this analysis indicate that two of the estimated concentrations fall below their respective TLV's, and the concentration of ammonium chloride is approximately equal to its TLV. Considering the conservative nature of the adjusted nonoccupational TLV on which the comparisons were made, none of the nonradioactive gases appear to be at concentrations that might pose a serious health hazard. The ammonia level is about 80 percent of the estimated "natural" background concentration and only about 10 percent of the average urban concentration (Table 6.19).

6.1.2.3 Trace Metals and Particulates in the Form of Dust

We identified seventeen trace metals and particulates in the form of dust as potential airborne emissions from uranium mines. Table 6.22 presents projected airborne concentrations of the metals and particulates at the site of the maximum individual for six mine classifications. As might

Table 6.20 Combustion product concentrations at the site of the maximum individual with comparisons, $\mu g/m^3$

Pollutant	Average underground mine	Average large underground mine	Average surface mine	Average large surface mine	Natural background concentration[a]	Average urban concentration[a]	Non-occupational TLV[b]
Particulates of combustion	1.4E-3	1.6E-2	9.7E-2	4.5E-1	NR[c]	NR	NR
SO_x	1.2E-2	1.3E-1	5.5E-1	2.2E+0	5E+0	6.2E+1	3.1E+1
CO	9.7E-2	1.1E+0	4.3E+0	1.8E+1	1.0E+2	7.0E+3	1.3E+2
NO_x	1.6E-1	1.8E+0	7.1E+0	3.0E+1	4.0E+1	1.4E+2	2.1E+1
Hydrocarbons	1.6E-2	1.8E-1	7.1E-1	3.1E+0	NR	5.0E+2	NR

[a] See Table 6.19.

[b] Nonoccupational TLV = TLV (mg/m^3) x 40 hr/168 hr x 10^{-2} x 10^3 $\mu g/mg$ (ACGIH76).

[c] NR - Not reported.

Table 6.21 A comparison of the airborne concentrations of nonradioactive gases at the hypothetical in situ leach site with threshold limit values

Contaminant	Atmospheric Concentration[a] ($\mu g/m^3$)	TLV[b] (mg/m^3)	Non-occupational[c] TLV ($\mu g/m^3$)	Percent of Nonoccupational TLV
NH_3	8.1	18	43	19
NH_4Cl	24	10	24	100
CO_2	60	9000	21,400	0.3

[a] Location of maximum individual.

[b] Source: ACGIH76.

[c] Nonoccupational TLV = TLV (mg/m^3) x 40 hr/168 hr x 10^{-2} x 10^3 $\mu g/mg$.

be expected, large surface mine emissions usually have the greatest concentrations, and those from inactive underground mines the least. Projected metal concentrations range from a low of about 5×10^{-7} $\mu gm/m^3$ of cobalt from inactive underground mines to a high of about 1 $\mu gm/m^3$ of potassium from large surface mines.

Table 6.23 shows where particulates (dust) or trace metal air concentrations are estimated to exceed natural background or average urban air concentrations (Table 6.19). Several trace metal air concentrations exceed "natural" background; however, only the estimated air concentration of particulates (dust) exceeds the air concentration of airborne pollutants in urban areas.

We evaluated the significance of these concentrations by comparing them with threshold limit values (TLV's) for workroom environments published by the American Conference of Governmental Industrial Hygienists (ACGIH76). These TLV's, which are for occupational workers and a 40-hour workweek, were adjusted by multiplying by 40/168 to convert them to continuous exposure values and dividing by 100 to make them applicable to the general public. Table 6.24 is a tabulation of the adjusted TLV's, the projected concentrations of metals and particulates (from Table 6.22), and the ratio of these concentrations to the adjusted TLV's. The sums of these ratios provide a measure of whether a mixture of the metals would be a significant problem, a sum greater than one indicating that the "composite" TLV has been exceeded.

Table 6.24 shows that in no case does a single metal exceed its TLV, nor do any of the mixtures exceed a "composite" TLV. Although TLV's were not available for potassium and strontium, their low toxicity and low concentrations make it unlikely that their addition to the sums would change this conclusion. For the worst case, large surface mines, the sum of ratios is only about 17 percent of the limit.

Particulates, on the other hand, present a different picture. The TLV for nonspecific particulates, nuisance dust, was chosen for comparison. It can be seen that the TLV is exceeded by a factor of six at the large model surface mine and nearly exceeded at the average model surface mine. About 50% of the exposure to dust is from vehicular traffic, and about 30% results from mining activities within the pit.

In summary, specific trace metal airborne emissions from uranium mines do not appear to present a significant hazard, either singly or as com-

Table 6.22 Stable trace metal airborne concentrations at the site of the maximum individual, $\mu g/m^3$

Trace metal	Avg. underground mine	Avg. large underground mine	Avg. surface mine	Avg. large surface mine	Inactive underground mine	Inactive surface mine
As	3.1E-5	1.9E-4	2.6E-4	1.5E-3	3.1E-6	1.5E-5
Ba	5.1E-4	1.8E-3	7.0E-3	4.2E-2	3.6E-5	1.6E-4
Co	4.0E-6	3.1E-5	1.1E-5	4.7E-5	4.5E-7	2.9E-6
Cu	3.3E-5	1.5E-4	4.4E-4	2.6E-3	2.2E-6	1.1E-5
Cr	5.0E-5	1.4E-4	1.1E-3	6.9E-3	8.9E-7	3.6E-6
Fe	9.7E-3	4.1E-2	1.4E-1	8.5E-1	6.3E-4	2.7E-3
Hg	7.2E-6	1.5E-5	1.8E-4	1.1E-3	NA[a]	NA
K	1.3E-2	6.3E-2	1.7E-1	1.0	9.8E-4	4.4E-3
Mg	9.4E-4	6.9E-3	2.5E-3	1.0E-2	1.3E-4	6.2E-4
Mn	7.1E-4	2.8E-3	1.1E-2	6.8E-2	3.7E-5	1.7E-4
Mo	3.3E-5	2.3E-4	1.4E-4	6.7E-4	4.5E-6	2.0E-5
Ni	4.9E-6	3.9E-5	1.4E-5	5.8E-5	8.9E-7	3.6E-6
Pb	4.1E-5	2.0E-4	5.4E-4	3.2E-3	3.1E-6	1.4E-5
Se	3.1E-5	2.2E-4	1.2E-4	5.9E-4	4.5E-6	1.9E-5
Sr	1.7E-4	5.5E-4	3.4E-3	2.1E-2	4.9E-6	2.3E-5
V	4.7E-4	3.0E-3	2.2E-3	1.8E-2	5.4E-5	2.5E-4
Zn	2.6E-5	9.6E-5	4.6E-4	2.8E-3	1.3E-6	5.2E-6
Part[b]	1.2	3.9	2.3E+1	1.4E+2	3.9E-2	1.7E-1

[a] NA - Not available.

[b] Part. - Particulates (dust).

Table 6.23 Comparison of stable trace metal airborne concentrations at the location of the maximum individual with natural background concentrations and average urban concentrations of these airborne pollutants

Exceed Natural Background[a]	Exceed Average Urban Concentration[a]
Average Large Surface Mine	
Ba, Cr (possible), Fe, Hg (possible), Mn, Mo, Pb, Sr, V, particulates	Particulates
Average Surface Mine	
Ba, Cr (possible), Mn, V	None
Average Large Underground Mine	
V	None
Average Underground Mine	
None	None

[a] See Tables 6.19 and 6.22.

Table 6.24 Comparison of trace metal airborne concentrations at the site of the maximum individual with threshold limit values (TLV's) in the workroom environment adjusted for continuous exposure to the general public, $\mu g/m^3$

Trace metal	Adjusted[a] TLV	Average Underground Mine		Average Large Underground mine		Average Surface Mine		Average Large Surface mine		Inactive Underground mine		Inactive Surface mine	
		Conc.	Conc./TLV	Conc.	Conc./TLV	Conc.	Conc./TLV	Conc.	Conc./TLV	Conc.	Conc./TLV	Conc.	Conc./TLV
As	1.2	3.1E-5	3E-5	1.9E-4	2E-4	2.6E-4	2E-4	1.5E-3	1E-3	3.1E-6	3E-6	1.5E-5	1E-5
Ba	1.2	5.1E-4	4E-4	1.8E-3	2E-3	7.0E-3	6E-3	4.2E-2	4E-2	3.6E-5	3E-5	1.6E-4	1E-4
Co	0.24	4.0E-6	2E-5	3.1E-5	1E-4	1.1E-5	5E-5	4.7E-5	2E-4	4.5E-7	2E-6	2.9E-6	1E-5
Cu	0.48	3.3E-5	7E-5	1.5E-4	3E-4	4.4E-4	9E-4	2.6E-3	5E-3	2.2E-6	5E-6	1.1E-5	2E-5
Cr	1.2	5E-5	4E-5	1.4E-4	1E-4	1.1E-3	9E-4	6.9E-3	6E-3	8.9E-7	7E-7	3.6E-6	3E-6
Fe	12	9.7E-3	8E-4	4.1E-2	3E-3	1.4E-1	1E-2	8.5E-1	7E-2	6.3E-4	5E-5	2.7E-3	2E-4
Hg	0.12	7.2E-6	6E-5	1.5E-5	1E-4	1.8E-4	2E-3	1.1E-3	9E-3	NA	----	NA	--
K	NA[b]	1.3E-2	---	6.3E-2	---	1.7E-1	--	1.0E+0	--	9.8E-4	----	4.4E-3	--
Mg	24	9.4E-4	4E-5	6.9E-3	3E-4	2.5E-3	1E-4	1.0E-2	4E-4	1.3E-4	5E-6	6.2E-4	3E-5
Mn	12	7.1E-4	6E-5	2.8E-3	2E-4	1.1E-2	9E-4	6.8E-2	6E-3	3.7E-5	3E-6	1.7E-4	1E-5
Mo	12	3.3E-5	3E-6	2.3E-4	2E-5	1.4E-4	1E-5	6.7E-4	6E-5	4.5E-6	4E-7	2.0E-5	2E-6
Ni	0.24	4.9E-6	2E-5	3.9E-5	2E-4	1.4E-5	6E-5	5.8E-5	2E-4	8.9E-7	4E-6	3.6E-6	2E-5
Pb	0.36	4.1E-5	1E-4	2.0E-4	6E-4	5.4E-4	2E-3	3.2E-3	9E-3	3.1E-6	9E-6	1.4E-5	4E-5
Se	0.48	3.1E-5	6E-5	2.2E-4	5E-4	1.2E-4	2E-4	5.9E-4	1E-3	4.5E-6	9E-6	1.9E-5	4E-5
Sr	NA	1.7E-4	--	5.5E-4	--	3.4E-3	--	2.1E-2	--	4.9E-6	--	2.3E-5	--
V	1.2	4.7E-4	4E-4	3.0E-3	2E-3	2.2E-3	2E-3	1.8E-2	2E-2	5.4E-5	4E-5	2.5E-4	2E-4
Zn	12	2.6E-5	2E-6	9.6E-5	8E-6	4.6E-4	4E-5	2.8E-3	2E-4	1.3E-6	1E-7	5.2E-6	4E-7
Total of ratios			2E-3		1E-2		3E-2		1.7E-1		2E-4		7E-4
Particulates:													
Dust	24[c]	1.2E+0	5E-2	3.9E+0	2E-1	2.3E+1	1E+0	1.4E+2	6E+0	3.9E-2	2E-3	1.7E-1	7E-3

[a] Adjusted TLV = Occupational TLV (mg/m^3) x 40 hr/168hr x 10^3 $\mu g/mg$ x 1/100.

[b] NA - Not available.

[c] Limit for nuisance dust - total mass.

Source: Workroom TLV's from ACGIH76.

posite mixtures, when evaluated against adjusted threshold limit values. However, particulate emissions, at least for surface mines, require further evaluation. If model predictions can be verified by measurement, control measures are indicated.

6.1.3 Radioactive Aquatic Emissions

We used the data on radioactive releases from mine dewatering (Sections 3.3.3 and 3.4.3) to estimate the public health impact of mining operations at a typical active underground mining site (New Mexico) and a typical active surface mining site (Wyoming). The health risks estimated in this section are of fatal cancers and genetic effects to succeeding generations. Dose equivalents and health risks per year of active mine operation are estimated for the maximum and average individuals and for the population of each assessment area. These calculated dose equivalents and health risk estimates are believed to be higher than the actual dose equivalents and health risks because of the conservative ass ptions required to predict movement of radionuclides in surface waters (see Section J.2 of Appendix J). Very few data are available on aquatic releases from inactive mines; hence, the significance of these releases, particularly for Colorado and Utah where inactive mines are numerous, could not be determined.

The individual and population dose equivalents presented in this section are computed using the models and parameters discussed in Appendix J. The health risk estimates are generated by the following procedures:

a. For inhalation or ingestion of radionuclides, the quantity of radionuclides taken into the body is determined as part of the dose equivalent calculations. This quantity is multiplied by a health risk per unit intake conversion factor.
b. For external irradiation from ground deposited radionuclides or from air submersion, the dose equivalents are calculated and multiplied by a health risk per unit dose equivalent conversion factor.

The health risk per unit intake and health risk per unit external dose equivalent conversion factors for aquatic releases are listed in Tables J.13 and J.14, Appendix J. This appendix also discusses the health risk assessment methodology used to obtain the risks presented in this section. Uranium and Ra-226 releases are given for both active mining sites. It is assumed that the stated uranium releases are entirely U-238 and that U-234 is in equilibrium with the U-238 but that Th-230 precipitates out of the mine water.

Table 6.25 Annual radiation dose equivalent rates due to aquatic releases from the New Mexico model underground mine

Organ	Maximum Individual Dose Rate (mrem/y)	Average Individual Dose Rate (mrem/y)	Population Dose Rate (person-rem/y)
Endosteal	5.6E+1	5.0	3.2E+2
Red Marrow	2.0	1.6E-1	1.1E+1
Lung	1.3	2.1E-3	1.4E-1
Liver	5.5E-1	2.9E-2	1.9
Stomach Wall	1.9E-1	3.8E-3	2.5E-1
LLI Wall[(a)]	9.4E-1	6.6E-2	4.3
Thyroid	4.5E-1	2.5E-2	1.6
Kidney	2.8E+1	2.4	1.6E+2
Muscle	4.9E-1	2.5E-2	1.6
Ovaries	4.1E-1	2.4E-2	7.8E-1
Testes	4.7E-1	2.4E-2	7.8E-1
Weighted Mean	2.2	1.5E-1	9.9

[(a)]Lower large intestine wall.

Table 6.26 Annual radiation dose equivalent rates due to aquatic releases from the Wyoming model surface mine

Organ	Maximum Individual Dose Rate (mrem/y)	Average Individual Dose Rate (mrem/y)	Population Dose Rate (person-rem/y)
Endosteal	6.8E-1	2.1E-1	3.4
Red Marrow	3.8E-2	7.4E-3	1.2E-1
Lung	2.3E-2	1.0E-4	1.7E-3
Liver	3.0E-2	2.8E-3	4.5E-2
Stomach Wall	1.0E-2	2.8E-4	4.6E-3
LLI Wall[(a)]	2.9E-2	7.7E-3	1.3E-1
Thyroid	1.8E-2	1.4E-3	2.3E-2
Kidney	4.0E-1	1.1E-1	1.8
Muscle	1.9E-2	1.5E-3	2.4E-2
Ovaries	1.5E-2	1.5E-3	1.2E-2
Testes	1.8E-2	1.4E-3	1.2E-2
Weighted Mean	4.0E-2	7.1E-3	1.2E-1

[(a)]Lower large intestine wall.

Table 6.27 Individual lifetime fatal cancer risk and committed fatal cancers to the population residing within the assessment areas

Source	Maximum exposed individual lifetime fatal cancer risk for operation of the mine		Average exposed individual lifetime fatal cancer risk for operation of the mine[a]		Committed fatal cancers for the assessment area population for operation of the mine	
	1 yr.	17 yrs.	1 yr.	17 yrs.	1 yr.	17 yrs.
Underground mine site (New Mexico)	3.3E-7	5.6E-6	2.0E-8	3.4E-7	1.3E-3	2.2E-2
Surface mine Site (Wyoming)	7.1E-9	1.2E-7	9.6E-10	1.6E-8	1.6E-5	2.7E-4

[a]The average individual risk is the cumulative population risk divided by the population residing within the assessment area.

Also, it is assumed that Rn-222, Pb-214, Bi-214, Pb-210, and Po-210 are in equilibrium with the Ra-226. For example, a reported release rate of 0.01 Ci/yr of U-238 would be reflected in the analyses as 0.01 Ci/yr of U-238 and 0.01 Ci/yr of U-234. In like manner, a release of 0.001 Ci/yr of Ra-226 would be reflected in the analyses as 0.001 Ci/yr Ra-226, 0.001 Ci/yr Rn-222, 0.001 Ci/yr Pb-214, 0.001 Ci/yr Bi-214, 0.001 Ci/yr Pb-210, and 0.001 Ci/yr Po-210.

The maximum individual, average individual, and population annual dose equivalent rates due to release of mine water containing radionuclides are given in Tables 6.25 and 6.26 for the two active uranium mine sites. The population dose equivalent rates are the sum of the dose equivalent rates to all individuals residing within the assessment areas due to the annual release from the model uranium mine. Average individual dose equivalent rates are computed by dividing the population dose equivalent rates by the number of persons in the assessment area.

The dose equivalent rates in Tables 6.25 and 6.26 indicate that the endosteal cells and kidney are the highest exposed target organs. Ingestion is the predominant exposure mode for both the endosteal cells and the kidney.

Individual lifetime fatal cancer risks and committed fatal cancers to the population within the assessment area for radionuclide releases due to mine dewatering are presented in Table 6.27. The maximum and average individual lifetime risks (columns 2 and 3, respectively) and the committed fatal cancers to the population within the assessment area (column 4) are shown for both one year of release of radionuclides due to mine dewatering and, in parenthesis, for the cumulative release over the 17 years of mine operation. To compute the 17-year risks, the one-year risks are multiplied by 17, which assumes equal annual radionuclide discharges. At both the model underground (New Mexico) and surface (Wyoming) mines, the majority of the risk is from releases of U-238, U-234, and Po-210.

A perspective on the additional fatal cancers estimated for the population (Table 6.27) can be gained by realizing that the probability of an individual dying of cancer of all types is 0.15 (Ba79). Taking the New Mexico assessment area (64,950 persons) as an example, the expected number of deaths from all forms of cancer for this population is 9,743 persons. For the 17 years of mine operations, the estimated increase in the number of deaths from cancer in the assessment area population is 0.022 deaths

(Table 6.27). This represents a 0.00023 percent increase in the expected fatal cancer occurrences in the assessment area population as a result of operation of the underground mine in New Mexico over its 17-year active life. For the Wyoming assessment area (16,230 persons), the estimated increase in the expected fatal cancer deaths due to operation of the surface mine for 17 years is 0.000011 percent.

Table 6.28 presents the genetic risks to succeeding generations, for exposure to both individuals and the population within the assessment area, caused by mine dewatering radionuclide releases. The genetic risks to succeeding generations of maximum and average exposed individuals (columns 2 and 3, respectively) and the committed genetic effects to the descendants of the present population within the assessment area (column 4) are shown for one year of releases. The mechanics and assumptions used to estimate the genetic effects are similar to those used to estimate fatal cancer risks (see Appendix J). For both the model underground (New Mexico) and surface (Wyoming) mines the majority of the risk is from releases of U-238, U-234, and Po-210.

The risks of additional genetic effects due to the discharge of contaminated mine water from model uranium mine sites are very small when compared to the normal occurrence of hereditary diseases. As given in Section 6.1.1, the natural incidence of genetic effects is 60,000 per million births (NAS72), or 0.06 effects per birth. This natural incidence rate is equivalent to 848 effects per year per million persons, considering a birth rate of 0.01413 births per person-year. Taking the New Mexico site as an example, the normal incidence of genetic effects for the assessment area population (64,950 persons) during the 17 years of operation of the mine would be 936 genetic effects. The increase in genetic effects committed to the assessment area population during the 17 years of operation is 0.015 genetic effects committed. Thus, the genetic effects committed due to aquatic wastes released during the operation of the New Mexico underground mine are only 0.0016% of the genetic effects which occur due to other causes during the mine operating life. For the Wyoming site (16,230 persons), the genetic effects committed due to aquatic wastes released during the operation of the model surface mine are only 0.0001% of the genetic effects which occur due to other causes during the mine operating life. It

Table 6.28 Genetic risks to succeeding generations of an individual and committed genetic effects to descendants of the present population residing within the assessment area

Source	Genetic effects committed to succeeding generations of an individual for operation of the mine for 1 year[a]		Genetic effects committed to the descendants of the present population for operation of the mine for 1 year
	Maximum Individual	Average Individual	
Underground mine site (New Mexico)	4.5E-7	3.3E-8	9.0E-4
Surface mine site (Wyoming)	1.4E-8	2.0E-9	1.4E-5

[a]Genetic effects assume 1 birth per person.

should be noted that genetic effect risks to descendants of individuals cannot be added to somatic effect risks for these individuals.

6.1.4 Nonradioactive Aquatic Emissions

Data on nonradiological emissions from uranium mines via the water pathway are limited. Table 6.29 presents available estimates of concentrations of four trace metals plus sulfate and suspended solids in discharge streams from the model surface mine located in Wyoming and seven trace metals plus sulfate and suspended solids from the model underground mine located in New Mexico. These concentrations are calculated after dilution in the first order tributaries (Appendix J) and represent average concentrations for the assessment areas. The concentrations presented in Table 6.29 are conservative since, with the exception of sulfates, loss of contaminants due to precipitation, adsorption, and infiltration to shallow aquifers are not considered. The concentrations are calculated by diluting discharges from a mine into the first order surface streams with no losses. For sulfate, a more realistic approach is taken since only 20 percent of it is assumed to remain in solution in the surface stream, as discussed in Section 3.3.3.1.4.

Also presented in Table 6.29 are recommended agricultural water concentration limits for livestock and irrigation for several of these elements (EPA73). Drinking water limits are not presented because public water supplies are normally derived from groundwater rather than surface water, so drinking water would not be a pathway of concern for the average individual in the assessment area. Though drinking water would be a potentially significant pathway for the maximum individual, the data available for this analysis did not allow a reliable prediction of groundwater concentrations due to mine dewatering (Appendix J). For this reason, the impact of nonradioactive waterborne emission on the maximum exposed individual could not be evaluated. The ratios of the average water concentrations to these limits are also listed in Table 6.29 and show that only molybdenum from the underground mine approaches its limit (irrigation). Also, the sums of the ratios being less than one indicate that mixtures of the metals would not exceed a "composite limit" for an average individual in the assessment area.

Table 6.29 Comparison of nonradiological waterborne emissions from uranium mines with recommended agricultural water quality limits

Parameter	Recommended Limits, mg/ℓ		Model Surface Mine			Model Underground Mine		
	Livestock	Irrigation	Avg. Water Conc., mg/ℓ	Ratio Avg./Livestock Limit	Ratio, Avg./ Irrigation Limit	Avg. Water Conc., mg/ℓ	Ratio Avg./Livestock Limit	Ratio, Avg./ Irrigation Limit
Arsenic	0.2	0.1	1.4E-4	0.0007	0.0014	3.1E-4	0.0016	0.0031
Barium	NA[a]	NA				2.0E-2		
Cadmium	0.05	0.01	1.1E-4	0.0022	0.011	1.6E-4	0.0032	0.016
Molybdenum	NA	0.01				7.0E-3		0.70
Selenium	0.05	0.02				1.6E-3	0.032	0.08
Zinc	25	2.0	4.8E-4	0.00002	0.00024	1.1E-3	0.00004	0.00055
Uranium	NA	NA	2.0E-3			3.5E-2		
Sulfate	NA	NA	4.9			2.9		
Total suspended solids	NA	NA	5.8E-1			6.8E-1		
Totals				0.0029	0.013		0.037	0.80 (0.1)[b]

[a] NA - Not available.

[b] Excluding molybdenum.

Because of the limited number of data available, it is difficult to evaluate the significance of these discharges. Although molybdenum could be a problem, it is not possible to quantify the risk from molybdenum to the maximum individual without having estimates of drinking water concentrations. Uranium, the metal estimated to be in highest concentration (Table 6.29), has no established limits based on chemical toxicity in the United States. In Canada, the maximum acceptable concentration for uranium in drinking water based on chemical toxicity has been set at 0.02 mg/ℓ (0.04 mg/day), considering a continuous lifetime intake rate of 2 liters of water per day (HWC78). It is reasonable to assume that limits for uranium in water used for irrigation and to water livestock would exceed the drinking water limit. Hence, based on the estimated uranium concentrations at surface (0.002 mg/ℓ) and underground (0.035 mg/ℓ) uranium mines, the water would probably be acceptable for irrigation and livestock watering. The other constituents, such as solids and sulfates, for which limits are not available, have minimal or no toxic properties.

It is premature to conclude the health hazard caused by non-radiological waterborne emissions from uranium mines. Before definitive conclusions can be reached, additional information is needed. Of particular interest would be data on water use patterns in the vicinity of the mines and the degree to which the mine discharges may infiltrate groundwater supplies.

6.1.5 Solid Wastes

6.1.5.1 Radium-226 Content

Solid wastes, consisting of sub-ore, waste rock, and overburden, at active and inactive uranium mines contain elevated concentrations of radium-226.* The sub-ore may contain as much as 100 pCi/g of radium-226. Even though the overburden and waste rock contain lower concentrations than the sub-ore, most of these wastes contain concentrations of radium-226 in quantities greater than 5 pCi/g (see Sections 3.3.1, 3.4.1, 3.7.1, and 3.7.2).

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* The radium-226 concentration in natural soil and rock is about 1 pCi/g.

Uranium mine wastes containing radium-226 in quantities greater than 5 pCi/g have been designated as "hazardous wastes" in a recently proposed EPA regulation (43FR58946, December 18, 1978) under the Resource Conservation and Recovery Act (RCRA). This is primarily due to the fact that the use of these wastes under or around habitable structures could significantly increase the chance of lung cancer to individuals occupying these structures.

6.1.5.2 Estimates of Potential Risk

We have estimated the risk of fatal lung cancer that could occur to individuals living in houses built on land contaminated by uranium mine wastes (Table 6.30). Risks were estimated for homes built on land containing radium-226 soil concentrations ranging from 5 to 30 pCi/g. The relationship between the indoor radon-222 decay product concentration and the radium-226 concentration in soil under a structure is extremely variable and depends upon many complex factors. Therefore, the data in Table 6.30 only illustrate the levels of risks that could occur to individuals living in structures built on contaminated land. These data should not be interpreted as establishing a firm relationship between radium-226 concentrations in soil and indoor radon-222 decay product concentrations.

Table 6.30 Estimated lifetime risk of fatal lung cancer to individuals living in homes built on land contaminated by uranium mine wastes

^{226}Ra in Soil (pCi/g)	Indoor Working Levels (WL)	Lifetime Risk of Fatal Lung Cancer[(a)] (per 100 persons)
5	0.02	2.5
10	0.04	5.0
20	0.08	10
30	0.12	15

(a) Based on an individual being inside the home 75 percent of the time.

The working level concentrations in Table 6.30 were derived from calculations made by Healey (He78), who estimated that 1 pCi/g of radium-226 in underlying loam-type soil would result in about 0.004 WL inside a house with an air change rate of 0.5 per hour. These calculated working levels are in reasonable agreement with measurements made by EPA (Fig. 6.1) at 21 house sites in Florida (S.T. Windham, U.S. Environmental Protection Agency, Written Communication, 1980). The Florida data were derived from the average radium-226 concentration in soil (core samples were taken to a maximum depth of three feet at each site) and the average radon-222 decay product concentration inside each structure.

6.1.5.3 Using Radium Bearing Wastes In The Construction of Habitable Structures

Wastes containing elevated levels of radium-226 have been used at a number of locations in the construction of habitable structures. In Grand Junction, Colorado, uranium mill tailings were widely used as landfill under and around the foundations of homes and other structures causing high radon-222 decay product concentrations inside many structures. To remedy this situation, Public Law 92-314 was passed in 1972 to establish a federal-state remedial action program to correct the affected structures. In Mesa County, Colorado, which includes Grand Junction, uranium mill tailings were identified at about 6,000 locations. About 800 of these locations are expected to receive corrective action because the radon decay product concentrations inside buildings constructed at these locations exceeded the remedial action criteria (DOE79). According to the criteria, dwellings and schoolhouses would be recommended for remedial action if the indoor radon decay product concentration exceeded 0.01 WL above background; other structures would be recommended for remedial action if the indoor radon decay product concentration exceeded 0.03 WL above background.

In central Florida, structures have been built on reclaimed phosphate land. The reclaimed land is composed of phosphate mining wastes that contain elevated radium-226 concentrations. EPA estimates that about 1,500 to 4,000 residential or commercial structures are located on 7,500 acres of the total 50,000 acres of reclaimed phosphate-mined lands (EPA79). A survey of 93 structures built on reclaimed phosphate land showed that about 40 percent of the structures had indoor radon-222 decay product concentrations in excess of 0.01 WL and about 20 percent had concentrations in

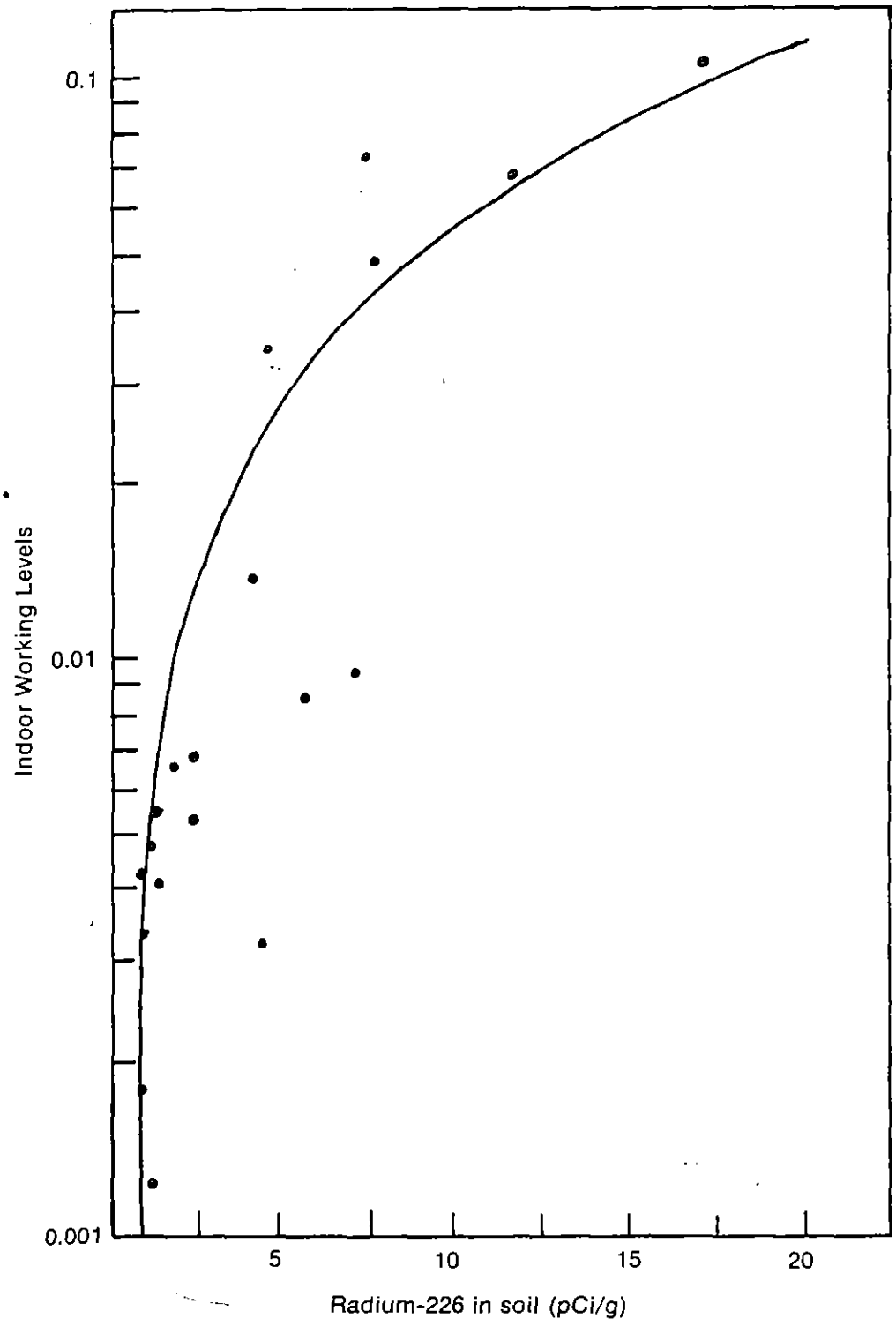

Figure 6.1 Average indoor radon-222 decay product measurements (in working levels) as a function of average radium-226 concentration in soil.

excess of 0.03 WL (EPA79). Lifetime residency in a structure with a radon-222 decay product concentration of 0.03 WL could result in twice the normal 3 to 4 percent risk of fatal lung cancer.

6.1.5.3.1 Use of Uranium Mine Wastes

We do not know to what extent the wastes from uranium mines have been removed from mining sites and used in local and nearby communities. However, while surveying in 1972 for locations with higher-than-normal gamma radiation in the Western States to locate uranium mill tailings material used in local communities, EPA and AEC identified more than 500 locations where "uranium ore" was believed to be the source of the elevated gamma radiation (ORP73). The specific type of ore (mill-grade, sub-ore, low-grade waste rock) was not determined as this was beyond the scope of the survey. At some locations, however, surveyors attempted to characterize the ore by using such terms as "ore spillage," "ore specimens," "low-grade crushed ore," or "mine waste dump material." Some locations were identified as sites of former ore-buying stations (ORP73).

Since it is unlikely that valuable mill-grade ore would have been widely available for off-site use, we suspect that uranium mine waste (perhaps sub-ore) may be the source of the elevated gamma radiation levels at many of the locations where large quantities of ore material are present. Table 6.31 shows the locations where higher-than-normal gamma radiation levels were detected during these surveys and the suspected sources of the elevated levels.

6.2 Environmental Effects

6.2.1 General Considerations

Minerals are necessary to augment man's existence and welfare; in order to obtain them, some form of mining is necessary. The very nature of mining requires disturbing the land surface, but may be considered transitory. To discuss the environmental effects of uranium mining in particular, it is convenient to divide the mining operations into three phases. The first phase includes the exploration for, and the delineation of, the ore body. This involves, in most cases, substantial exploratory and development drilling. The second phase involves the preparation of the mine site and the mining process itself. This phase includes the construction of service areas, dewatering impoundments, and access roads, digging or drilling of mine entries, etc. During the actual mining process, waste

Table 6.31 Gamma radiation anomalies and causes

Location	Number of Anomalies Detected	Cause of Anomaly				
		Tailings	Uranium Ore	Radioactive Source	Natural Radioactivity	Unknown
Arizona						
Cane Valley[a]	19	15	4			
Cameron	3		1			2
Cutter	5		4	1		
Tuba City	17	7			3	7
State Total	44	22	9	1	3	9
Colorado[b]						
Cameo	3	1				2
Canon City	187	36	24		99	28
Clifton	1083	159	31	3	14	876
Collbran	145	4	2			139
Craig	86	8	7		46	25
Debeque	109	2			1	106
Delta	43	1	3		29	10
Dove Creek	83	59	17	2	2	3
Durango	354	118	18	49	67	102
Fruita	1276	58	47	1	26	1144
Gateway	17	12	1	1		3
Glade Park	1	1				
Grand Valley	110	10	2			98
Gunnison	47	3	8	1	28	7
Leadville	91	18	2		65	6
Loma	199	10	3	1	4	181
Mack	90	6	2	1		82
Mesa	123	1	1			120
Mesa Lakes	3					3
Molina	43					43
Naturita	33	10	15	5	1	2
Nucla	13	3	6		2	2
Palisade	939	107	36	3	14	779
Plateau City	28	1				27
Rifle	810	168	20	7	1	614
Salida	64	6	2		52	4
Slick Rock	9	3	5	1		
Uravan	209	208				1
Whitewater	55		4		2	49
State Total	6253	1013	256	75	453	4456
Idaho						
Idaho City	3				2	1
Lowman	12	9			3	
Salmon	77	1	2		65	9
State Total	92	10	2		70	10

Table 6.31 (continued)

Location	Number of Anomalies Detected	Cause of Anomaly				
		Tailings	Uranium Ore	Radioactive Source	Natural Radioactivity	Unknown
New Mexico						
Bluewater	2	1	1			
Gamerco	5				5	
Grants	101	7	49	1	25	19
Milan	41	5	23	4	1	8
Shiprock	9	8	1	0		
State Total	158	21	74	5	31	27
Oregon						
Lakeview	18		2		10	6
New Pine Creek	4		1			3
State Total	22		3		10	9
South Dakota						
Edgemont	55	43	2	1	1	8
Hot Springs	45		3		17	25
Provo	4	3		1		
State Total	104	46	5	2	18	33
Texas						
Campbellton	7		1		6	
Coughran	1				1	
Falls City	5	2			3	
Fashing	1		1			
Floresville	16				14	2
George West	10				10	
Karnes City	10	2			6	2
Kenedy	22	1	1		13	7
Panna Maria	3				3	
Pawnee	1		1			
Pleasanton	21		1	2	17	1
Poth	15				14	1
Three Rivers	5	1			2	2
Tilden	11				11	
Whitsett	1				1	
State Total	129	6	5	2	101	15

Table 6.31 (continued)

Location	Number of Anomalies Detected	Cause of Anomaly: Tailings	Uranium Ore	Radioactive Source	Natural Radioactivity	Unknown
Utah						
Blanding	38	10	21		3	4
Bluff	2		1			1
Cisco	2		2			
Crescent Junction	2		1			1
Green River	23	1	14		1	7
Magna	27	1	1	1	21	3
Mexican Hat	5		4	1		
Mexican Hat (Old Mill)	14	10	1	2	1	
Moab	125	15	76	7	6	21
Monticello	59	31	16	3		9
Salt Lake City[c]	225	70	10	5	76	64
Thompson	30	26	3	0		1
State Total	552	164	150	19	108	111
Washington						
Creston	3				3	
Ford	1				1	
Reardan	10				10	
Springdale	2				2	
State Total	16				16	
Wyoming						
Hudson	8		2		5	1
Jeffery City	28	13	9	1	3	2
Lander	86	4	8	1	53	20
Riverton	86	15	14	1	33	23
Shirley Basin	9	9				
State Total	217	41	33	3	94	46
Totals	7587	1323	537	107	904	4716

[a] From EPA report ORP/LV-75-2, August 1975. Cane Valley was not included in initial gamma survey program.

[b] Excluding Grand Junction where non-tailings anomalies were not sub-categorized according to source.

[c] Salt Lake City was not completely surveyed.

Source: ORP73.

piles are produced, mine vents drilled or reamed, and pits opened and sometimes closed. In the third or retirement phase, the site is subject to deterioration from weathering ad infinitum. The extent of the deterioration depends somewhat on the amount and quality of reclamation conducted during this phase.

6.2.2 Effects of Mine Dewatering

Both surface and underground mines are dewatered in order to excavate or sink shafts and to penetrate and remove the ore body. Dewatering is by ditches, sumps, and drill holes within the mine or by high capacity wells peripheral to the mine and associated shafts. Dewatering rates up to 4 x 10^5 m^3/day have been reported in the literature. Average discharge for the surface and underground mines modeled herein are 3.0 and 2.0 m^3/minute/mine, respectively. Between 33 and 72 new mines are projected in the San Juan Basin of New Mexico alone. Total annual discharge is expected to exceed 1.48 x 10^9 m^3. Calculated effects include decreased flow in the San Juan (0.05 m^3/min) and the Rio Grande (0.85 m^3/min) rivers. Future mining will be primarily underground and the average mine depth will increase 275 percent, i.e., from 248 m to 681 m. Average mine discharge is expected to increase from 2.42 m^3/min to 13.8 m^3/min.

Aside from the hydraulic and water quality effects of discharging copious quantities of mine water to typically ephemeral streams, dewatering impacts are receiving increasing scrutiny because of the observed and calculated impacts on regional water availability and quality. Declines of water levels in regionally-significant aquifers of New Mexico and reduced base flow to surface streams are expected. Water quality effects relating to inter-aquifer connection and water transfer as a result of both dewatering and exploratory drilling have not been evaluated in any uranium mining area. In several Texas uranium districts, the effects of massive dewatering associated with surface mining are beginning to receive attention, but definitive studies have not yet begun and regulatory action is not expected in the near future. With respect to in situ leach mining, dewatering is not necessary and hence is not a concern. There is, however, some question concerning the practice of pumping large volumes of groundwater to restore aquifers. It is likely that both dewatering and aquifer restoration practices will come under increasing State regulation in water-short areas, particularly in areas of designated groundwater basins or where aquifers connect with fully-appropriated surface streams. The uncertainties surrounding environmental impacts of mining in this area can be

expected to increase, and additional, comprehensive investigations of the effects of mine dewatering and wastewater discharge are needed. Expansion in Wyoming and Texas surface and in situ leaching operations is similar, and these areas should be included in future investigations.

Uranium in water removed from mines through deliberate pumping or gravity flow is extracted for sale when the concentration is 2 to 3 mg/ℓ or more. If there is subsequent discharge to surface water, radium-226 is also removed down to concentrations of 2 to 4 pCi/ℓ to comply with NPDES permit conditions. Use of settling ponds at the mines also reduces total suspended solids and may reduce other dissolved constituents as a result of aeration and coprecipitation. Seepage from such settling ponds is believed to be low and, therefore, environmentally insignificant relative to groundwater. Management of waterborne solid wastes is inconsistent from one mine to another. In some cases, the solids are collected and put in with mill tailings, but in most cases they remain at the mine portal and are covered over.

For surface versus underground mines, we recognize certain inconsistencies in the parameters chosen to calculate contaminant loading of streams. Contaminant loadings from a model surface uranium mine were calculated for uranium, radium, TSS, sulfate, zinc, cadmium, and arsenic. As noted in Section 3.3.1, molybdenum, selenium, manganese, vanadium, copper, zinc, and lead are commonly associated with uranium deposits; however, there were too few data for the latter elements to develop an "average" condition. In addition, barium, iron, and magnesium can be abundant in New Mexico uranium deposits. There were insufficient data for these elements in the case of surface uranium mines in Wyoming, hence contaminant loadings were not calculated. Regional differences dictate which parameters are monitored for baseline definition and NPDES purposes. Not all potential contaminants are important in every region. For this reason and others, State and industry monitoring programs are inconsistent with respect to parameters. Since the scope of this study did not permit extensive field surveys, maximum reliance was placed on published, readily-available data.

In terms of parameters and concentrations, NPDES permit limits are inconsistent from one EPA Region to another and from one facility to another in a given Region. In part, this reflects previous screening of the effluent discharge data and natural variations in the chemistry of ore bodies.

However, the inconsistencies in parameters included and concentration limits are sufficiently large as to suggest reevaluating NPDES permits and specifying more consistent limits that more closely reflect contaminant concentrations and volumes of mine discharge.

Infiltration of most of the mine discharge in Wyoming and New Mexico is confirmed by field observations from these States. The modeling results agree with these field data. Furthermore, the modeling results, i.e., maximum infiltration, are consistent with those in the generic assessment of uranium milling (NRC79). Potable aquifers are defined under the Safe Drinking Water Act as those which contain less than 10,000 mg/ℓ TDS. Shallow groundwater throughout the uranium regions of the U.S. meets this criterion.

Considering that essentially all of the mine effluent infiltrates and is a source of recharge to shallow potable aquifers, NPDES limits should be influenced by the drinking water regulations and ambient groundwater quality. The latter is essentially never considered with respect to mine discharges. Extensive use of soils in both the saturated and unsaturated zones as sinks for significant masses of both water and toxic chemical constituents originating in the mine discharge necessitates further evaluation of the fate of these elements. Present understanding of fractionation and resuspension processes affecting stable and radioactive trace elements greatly limits accurate prediction of health and environmental effects of mine discharge.

6.2.3 Erosion of Mined Lands and Associated Wastes

Increased erosion and sediment yield result from mining activities ranging from initial exploration through the postoperative phase. Access roads and drilling pads and bare piles of overburden/waste rock and sub-ore constitute the most significant waste sources. Dispersal is by overland flow originating as precipitation and snowmelt. To a lesser extent, wind also transports wastes and sub-ore to the offsite environment. Underground mining is much less disruptive to the surface terrain than is surface mining. Documentation of the processes and removal rates is scarce and consists of isolated studies in Texas, Wyoming, and New Mexico. Conservatively assuming that sediment yields characteristic of the areas containing the mines also apply to the mine wastes, yields of overburden, waste rock, ore, and sub-ore amount to 90,000 m^3 per year. Total sediment

yield from all mining sources, including exploration and development activities, is estimated at 6.3×10^6 m^3.

Actual erosion rates from specific sources could be considerably above or below this value owing to such variables as pile shape and slope, degree of induration and grain size, vegetative cover, and local climatic patterns and cycles. Slope instability does present serious uranium mine waste problems throughout the mountainous uranium mining areas of Colorado (S.M. Kelsey, State of Colorado, written communication, 1979). Field observations in four western states confirm that some erosion characterizes essentially every pile but that proper reclamation, particularly grading and plant cover, provides marked improvement and may actually reduce sediment loss to below pre-mining levels. Unstabilized overburden, waste rock, and sub-ore piles revegetate rather slowly, even in areas of ample rainfall such as south Texas.

Stable trace metals such as molybdenum, selenium, arsenic, manganese, vanadium, copper, zinc, and lead are commonly associated with uranium ore and may cause deleterious environmental and health effects. Mercury and cadmium are rarely present. There is no apparent relationship between the concentration of trace metals and ore grade. In New Mexico ores, selenium, barium, iron, potassium, magnesium, manganese, and vanadium are most abundant. Presently, very few data are available to characterize the trace metal concentrations in overburden rock. Results of trace metal analyses of a few grab samples from several uranium mines in New Mexico and one in Wyoming show that except for selenium, vanadium, and arsenic, no significant trend attributable to uranium mining was present (N.A. Wogman, Battelle Pacific Northwest Laboratory, Written Communication, 1979). Considering the background concentration for these elements and the limited number of analyses, the inference of offsite contamination based on these elements is indefinite.

Ore storage piles, used to hold ore at the mine for periods averaging one month, are potential sources of contamination to the environment via dusts suspended and transported by the wind, precipitation runoff, and Rn-222 exhalation--all of which can be significantly reduced by proper management. Similarly, spoil piles remaining as a result of overburden, waste rock, or sub-ore accumulations left on the land surface after mining constitute a source of contaminants for transport by wind and water. Waste particles enriched in stable and radioactive solids and Rn-222 can be

transported by wind and precipitation runoff. Such transport can be reduced through proper grading and installation of soil covers protected by vegetation or rip-rap.

Soil samples collected from ephemeral drainage courses downgrade from inactive uranium mines in New Mexico and Wyoming generally revealed no significant offsite movement of contaminants (See Appendix G). For the New Mexico mines studied, Ra-226 was elevated to about twice local background at distances of 100 to 500 meters from the mine. Water and soil samples from a surface mining site in Wyoming showed no significant offsite movement of mine-related pollution although some local transport of stockpiled ore was evident in drainage areas on and immediately adjacent to one mine pit. The strongest evidence that mine wastes are a source of local soil and water contamination is the radiochemical data and uranium in particular. Substantial disequilibrium between radium and uranium may indicate leaching and remobilization of uranium, although disequilibrium in the ore body is also suspect.

6.2.4 Land Disturbance from Exploratory and Development Drilling

About 1.3×10^6 exploratory and development drill holes have been drilled through 1977 by the uranium mining industry (see Section 3.6.1). Using the estimated land area of 0.51 hectares disturbed per drill hole (Pe79), about 6.5×10^5 hectares of land have been disturbed by drilling through 1977. To further refine the estimates of land areas disturbed, we reviewed some recent drilling areas at three mine sites. From observing 187 recent drill sites, it was concluded that 0.015 ± 0.006 hectares per drill pad were physically disturbed. The error term for the estimates is at the 95 percent confidence level. The land area disturbed by roads to gain access to the drill sites was also estimated from aerial photography and amounts to 0.17 ± 0.11 hectares. The error term for this estimate is also at the 95 percent confidence level. The total area disturbed per drill site (drill pad and access roads) is 0.19 ± 0.11 hectares. Using the latter estimates from aerial photography, the total land area disturbed from all drilling through 1977 ranges from about 1000 to 4000 km^2 with a mean of about 2500 km^2. Drilling wastes removed from the boreholes can disturb additional land areas through wind and water erosion. Ore and sub-ore remaining in the drilling wastes can, in a radiological sense, disturb land areas around the drill site from erosion. The extent of the

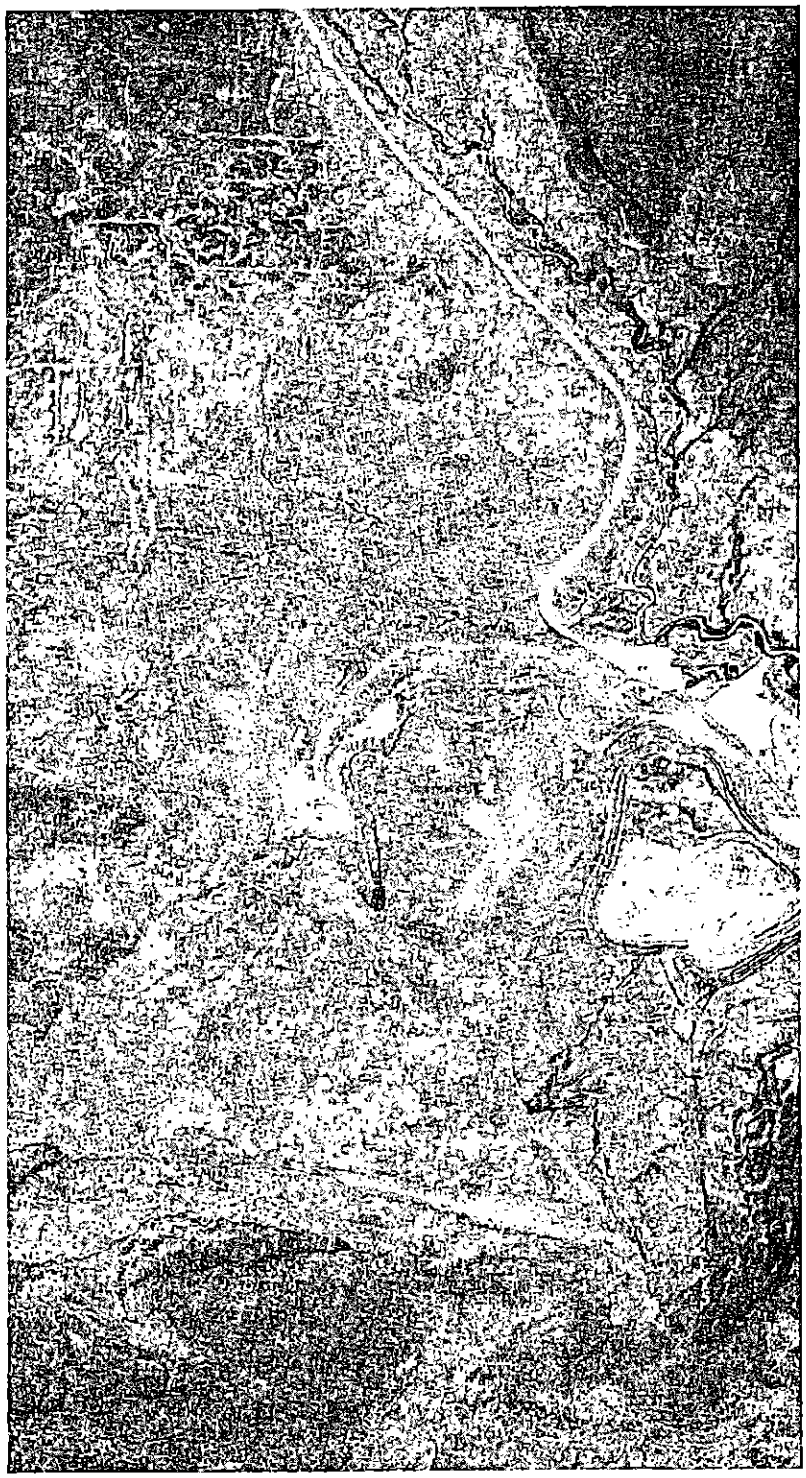

Figure 6.2 Example of natural reclamation of drill sites.

radiological contamination at drill sites is not known and cannot presently be estimated.

Some reversal of the initial environmental damage at older drill sites was also observed from aerial photographs. Figure 6.2 contains a typical medium-to-large surface uranium mine and some adjacent drilling areas that show the effects of weathering. New drill sites are in the upper left-hand corner of the photograph. The access roads and drill pads are plainly visible. It also appears that exposed drilling wastes remain at the drill site. The area left of center in the photograph shows drill sites that are probably intermediate in age. The drilling wastes remaining have very little voluntary vegetation growing on them, and appear to have been subject to wind erosion. Weathering of the drill pads and access roads is obvious, as they are hardly discernible. It appears, in these cases, that weathering may be considered a natural reclamation phenomenon. Old drill holes are located in the lower left corner of the photograph. The drilling wastes appear to be isolated dots; the drill pads and roads are almost indistinguishable from the surrounding terrain. It appears that weathering and volunteer plant growth tend to obscure scarring caused by roads located in relatively level areas. In Figure 6.3, an underground mine site, the access roads to the adjacent drill sites required extensive excavation because of the topography. These more severe excavation "scars" will probably remain for a long period of time.

In summary, the average number of drill holes per mine can be estimated by dividing the total number of holes drilled through 1977 by the number of active and inactive mines in existence in 1977:

$$\frac{1.3 \times 10^6 \text{ drill holes}}{3300 \text{ mines}} \approx 400 \frac{\text{drill holes}}{\text{mine}}. \qquad (6.1)$$

The total land area physically disturbed from drilling per mine is

$$400 \frac{\text{drill holes}}{\text{mine}} \times \frac{0.19 \text{ hectares}}{\text{drill hole}} \times \frac{\text{km}^2}{100 \text{ hectares}} = \frac{0.76 \text{ km}^2}{\text{mine}}. \qquad (6.2)$$

In some instances, weathering and volunteer plant growth (natural reclamation) tend to restore the land areas disturbed by drilling. In others, especially on rugged topography where extensive excavation has occurred, weathering may promote extensive erosion rather than natural reclamation. Any ore or sub-ore remaining at the drill sites is subject to erosion.

6.2.5 Land Disturbance from Mining

6.2.5.1 Underground Mines

At underground mines, some land area must be disturbed to accommodate equipment, buildings, wastes, vehicle parking, and so on. The disturbed area may range widely between mines in the same area or in different geographical areas. The land area disturbed by 10 mines was estimated from aerial photographs. Nine of the mines were in New Mexico and one was in Wyoming. The disturbed land area averaged 9.3 hectares per mine site and ranged from 0.89 to 17 hectares. Access roads for each mine site consumed about 1.1 hectares on the average and ranged from 0.20 to 2.59 hectares. Subsidence or the collapse of the underground workings also causes some land disturbances. An estimated 2.8 km^2 of land has subsided as a result of uranium mining in New Mexico from 1930-71 (Pa74). A crude estimate of the land disturbed from subsidence per mine can be made by dividing the subsided area by the number of inactive underground mines in New Mexico. This amounts to about 1.5 hectares per mine. The total area (mine site, access roads, and subsidence) disturbed by an underground mine is estimated to be 12 hectares.

6.2.5.2 Surface Mines

An estimate of land disturbed from surface mining was also made from aerial photographs of eight mining sites in New Mexico and two in Wyoming. The area estimates are for a single pit or a group of interconnected pits, including the area covered by mine wastes. The average disturbed area was estimated to be about 40.5 hectares and ranged from 1.1 to 154 hectares. Access roads for the pits averaged 2.95 hectares (0.03 km^2) and ranged from 0.18 to 18 hectares. The total area disturbed per mine site is about 44 hectares.

6.2.6 Retirement Phase

The actual exploration and mining of the uranium ore constitutes a very small portion of the total existence time of a mine when considered over a large time frame. The natural forces of erosion and weathering, as well as plant growth, will eventually change any work or alterations that man has made on the landscape. For example, underground mines may eventually collapse and fill with water if they are in a water table; waste piles erode and disperse in the environment; the sharp edges of pits become

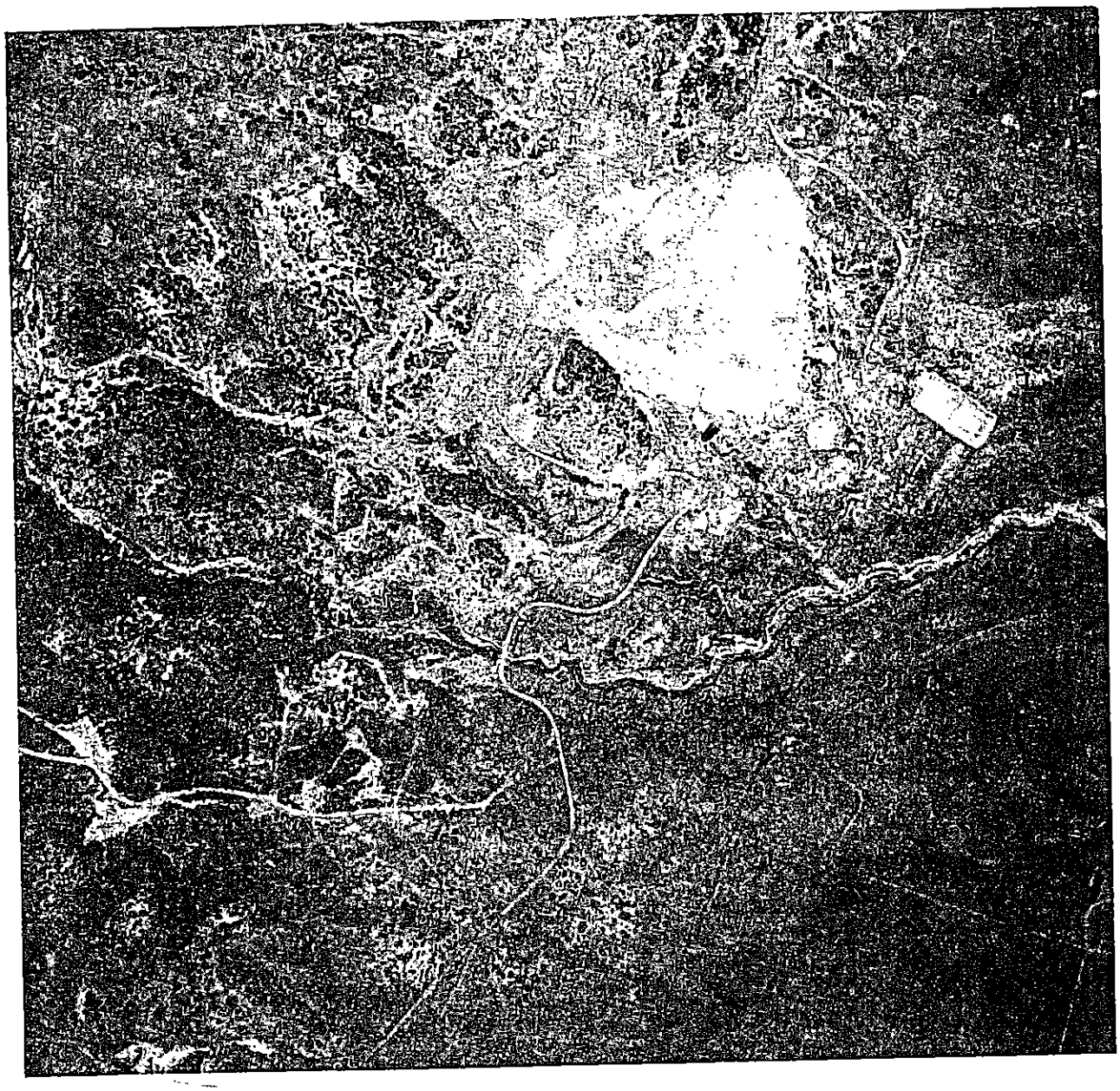

Figure 6.3 Inactive underground mine site.

smooth from wind and water erosion; lakes that are produced in pits fill up with sediment; vents and mine entries collapse, etc.

Perhaps one of the more important considerations associated with allowing a mine site to be naturally reclaimed is the dispersal of the mine wastes. Their removal from underground and subsequent storage on the surface constitute a technological enhancement of both radioactive materials and trace metals, creating a low-level radioactive materials disposal site. It appears that containment of the wastes would be preferred over their dispersal. Wastes from underground mines deposited near the entries are subject to substantial erosion. Figure 6.3 is an aerial photograph of an inactive underground uranium mine. The large light area is the waste pile and the small pile nearby is a heap-leach area. Erosion is occurring on both. A possible solution to this problem is to minimize the amount of wastes brought to the surface by backfilling mined-out areas. Another technique to minimize the dispersal of wastes into the environment by containment is to stabilize them. Unfortunately, a substantial quantity of wastes from past mining activities have been dumped in depressions and washes, which, in essence, enhances their dispersion into the environment. In retrospect, the wastes should have been stored in areas where minimal erosion would occur and then covered with sufficient topsoil to promote plant growth.

In surface mining, radiological containment can be accommodated by keeping the topsoil, waste rock, and sub-ores segregated during their removal. When backfilling, the materials can be returned to the pit in the order they were removed or in an order that would enhance the radiological quality of the ground surface. In this manner, the wastes would be contained and essentially removed from the biosphere. Figure 6.4 shows some examples of inactive and active surface mines. Some weathering and natural revegetation are noticeable around the inactive pits. Revegetation, on the other hand, appears to be relatively sparse at other inactive pits.

Erosion in inactive mining areas in New Mexico and Texas can result in deep gullying of mine waste and overburden piles. The mine wastes blanketing the foreground of Figure 6.5 are incised by an ephemeral stream that has been subsequently crossed by a roadbed in the immediate foreground. This particular mine, located in the Mesa Montanosa area immediately south of Ambrosia Lake, New Mexico, was active from 1957 to 1964. Thus, erosion occurred in about 15 years. In the background is a large mine waste pile,

the toe of which is being undercut by the same ephemeral stream (Fig. 6.6). No deliberate revegetation of the mine wastes dumped in either discrete piles or spread over the landscape (Fig. 6.7) is occurring, due in large part to the unfavorable physical and chemical characteristics of the wastes. The wastes are devoid of organic matter and are enriched in stable and radioactive trace elements, some of which are toxic to plant life. Low rainfall and poor moisture retention characteristics further suppress vegetative growth. As shown in Fig. 6.7, there is a sharp contrast between the vegetative cover on mine wastes versus that on the undisturbed rangeland in the background. Waste rock from many if not most of the mines in New Mexico, Utah, and Colorado is weakly cemented sandstone with numerous shale partings. Physical breakdown to loose, easily-eroded soil unsuitable for plant life is common (Fig. 6.8), and transport by overland flow and ephemeral streams occurs both during and long after the period of active mining (Fig. 6.9).

Depending on the degree of reclamation, if any, inactive surface mines in Texas vary considerably in the degree of erosion and revegetation. For example, the deep gullying shown in Fig. 6.10 developed in a period of one year. The mine wastes in this case were not contoured or covered to minimize gamma radiation, excessive erosion, or revegetation. In fact, the wastes were disturbed and shifted very recently in the course of constructing the holding pond (for mine water pumped from an active mine to the right of the picture) in the background. Drainage in this instance is internal, i.e., to a holding pond. In the background are more recent mine waste piles also showing deep gullying, scant vegetation, and lack of protective soil covering. Mine wastes in Texas are not completely returned to the mine primarily because of the excessive cost. As in the case of most mining operations, the bulking factor makes it physically impossible to completely dispose of the wastes in the mines.

Surface mines in Texas, particularly the older ones, also have associated overland flow to the offsite environment. Shown in Fig. 6.11 is a principal channel floored by unstabilized mine wastes and draining toward nearby grazing lands. Numerous deer and doves also were observed in the area and are actively pursued by sportsmen. The unstabilized mine in this photograph was last active several years ago, but most activity stopped in 1964. Vegetation has been very slow to reestablish and is essentially limited to a very hardy, drought-resistant willow shown in the center of the picture.

Figure 6.4 Example of active and inactive surface mining activities.

Figure 6.5 Mine wastes eroded by ephemeral streams in the Mesa Montanosa area, New Mexico.

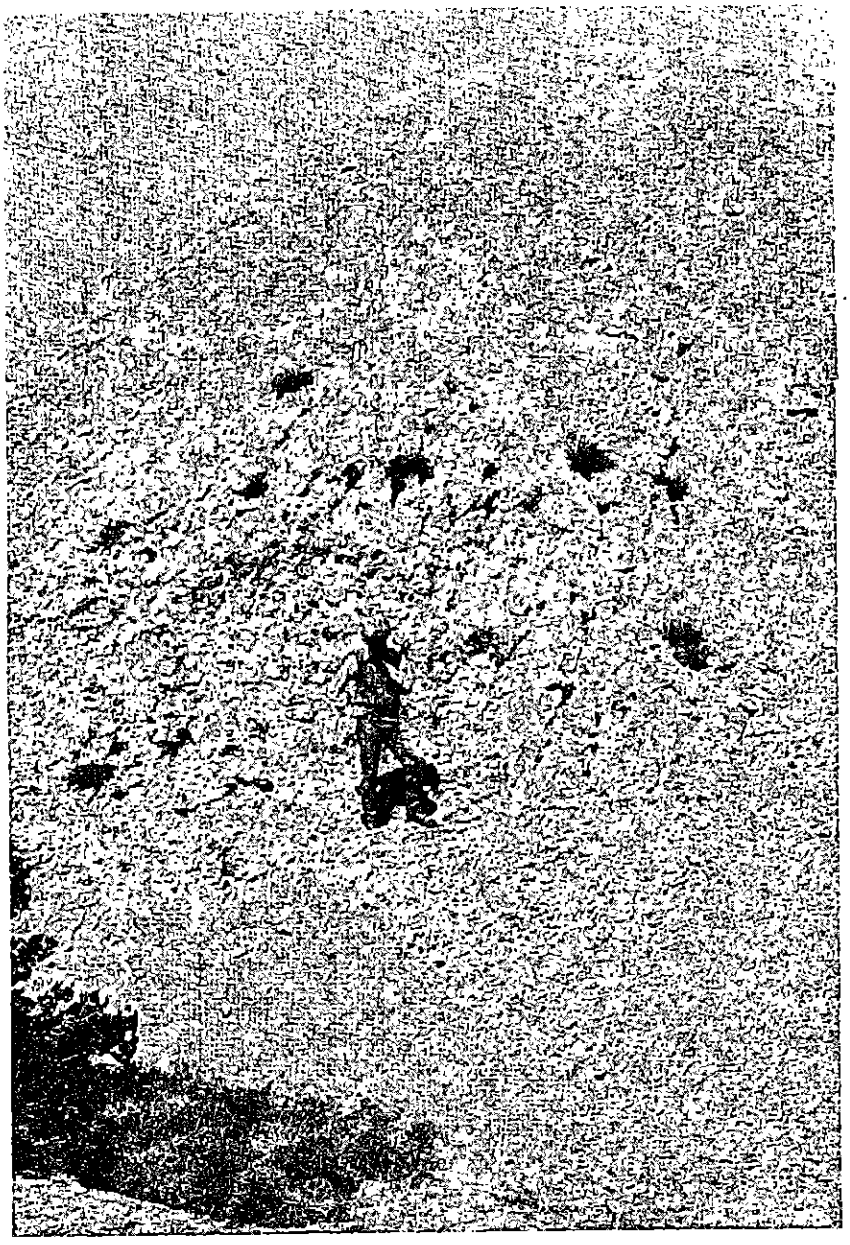

Figure 6.6 Basal erosion of a uranium mine waste pile by an ephemeral stream in the Mesa Montanosa area, New Mexico.

Figure 6.7 Scattered piles of mine waste at the Mesa Top Mine, Mesa Montanosa, New Mexico. Note the paucity of vegetation. Columnar object in background is a ventilation shaft casing.

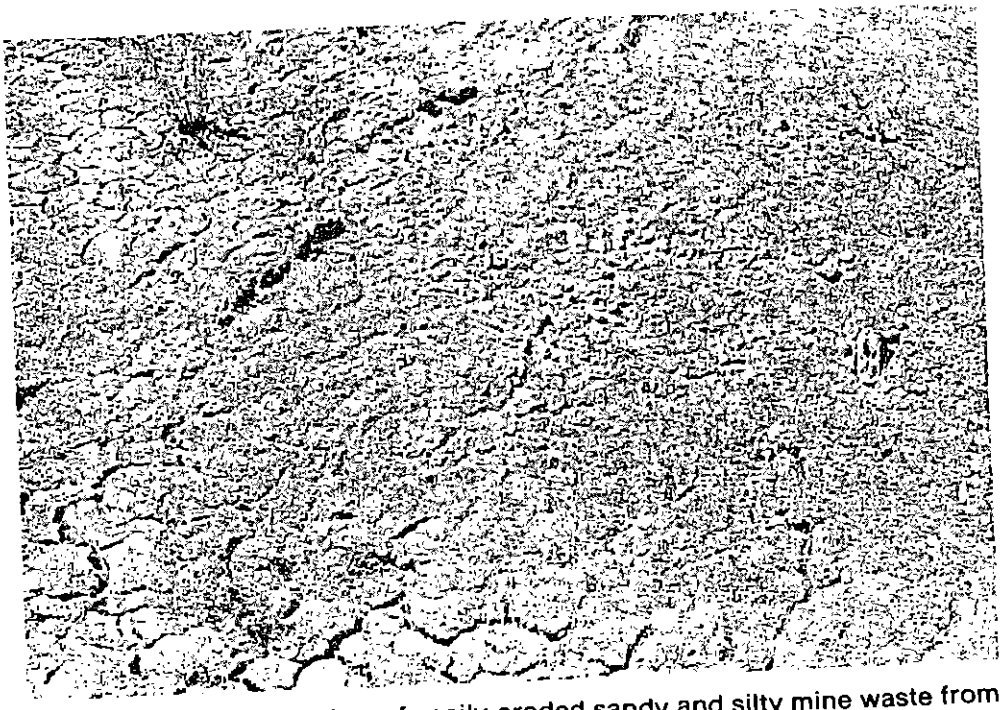

Figure 6.8 Close-up view of easily eroded sandy and silty mine waste from the Mesa Top Mine, Mesa Montanosa, New Mexico.

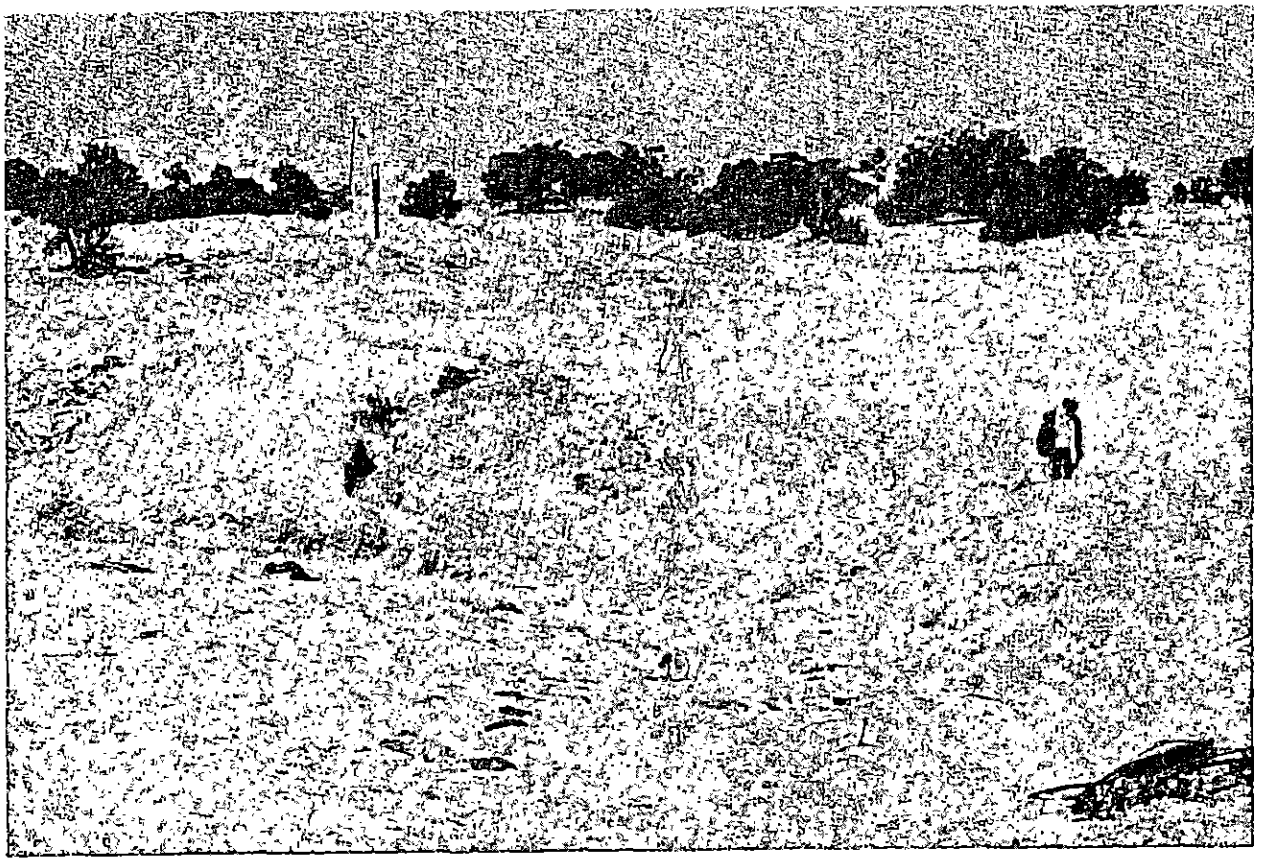

Figure 6.9 Gullying and sheet erosion of piled and spread mine wastes at the Dog Incline uranium mine, Mesa Montanosa, New Mexico.

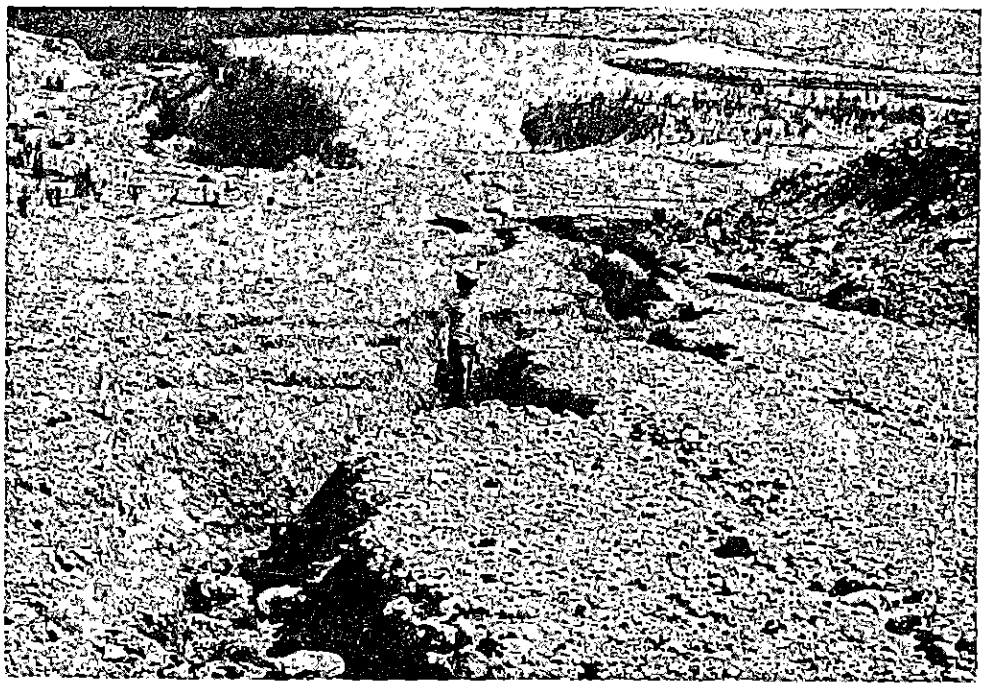

Figure 6.10 Recent erosion of unstabilized overburden piles at the inactive Galen mine, Karnes County, Texas.

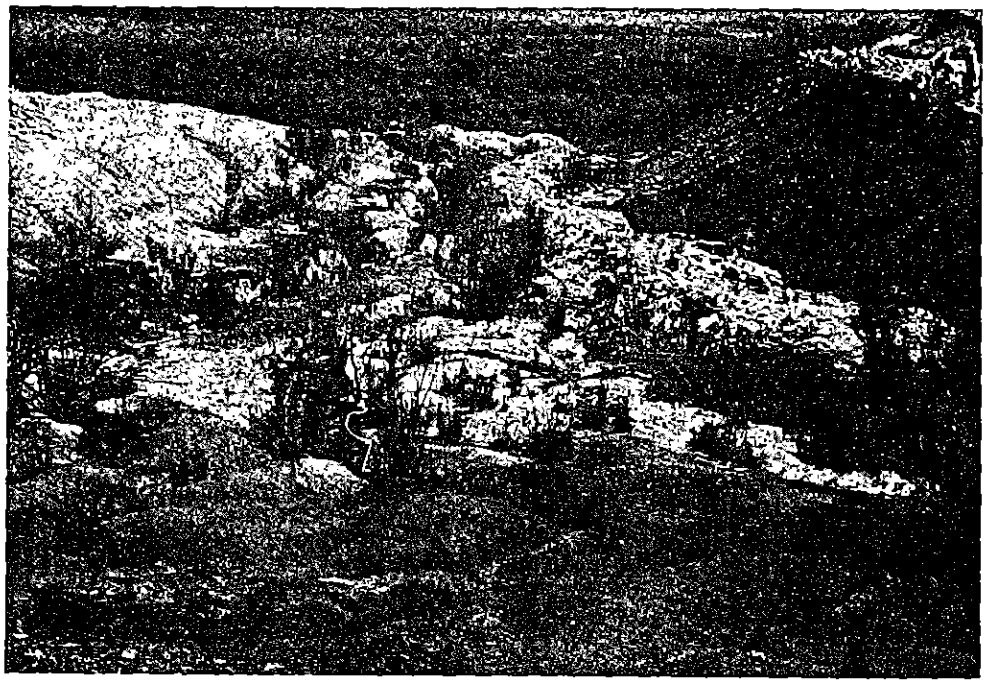

Figure 6.11 Unstabilized overburden piles and surface water erosion at the Galen Mine, Karnes County, Texas.

Mines stabilized within the last few years feature improved final contouring and use of topsoil and seeding to stimulate revegetation. The reclaimed spoil piles are then available for grazing. Because backfill cannot be complete (due to economic and bulking factors), part of the mine pit remains as shown in Figs. 6.12 and 6.13, which are of the same mine. The aerial view shows extensive patches of light colored soil devoid of vegetation. Here topsoil is missing and revegetation is minimal despite the 5 years elapsed since mining. Figure 6.13 is a closeup of one portion of the mine showing deep gullying, a thin layer of dark topsoil over relatively infertile sand and silt, and the vertical mine walls. Excavations like this must be fenced. They are a hazard to livestock and people. It is likely that erosion will continue to spread away from the mine; but the rate and consequence is unknown.

Although a mine site can be reclaimed to produce an acceptable aesthetic effect, it may not be suitable in a radiological sense. At the conclusion of surface mining, the remaining pit will contain exposed sub-ore on some of the pit walls and pit floor. Because most mines at least partly fill with water and the ore zone is thereby covered, gamma radiation and radon diffusion should be markedly reduced. Although water accumulation in the pit would be expected to have elevated concentrations of trace metals and radioactive materials, this condition would probably be temporary because of the eventual covering of the pit by sedimentation from inflow of surface water and materials sloughed from the pit walls. The natural reclamation process could be enhanced by tapering the pit walls to a more gradual slope and depositing the materials on the pit floor. If sub-ores are allowed to remain near the surface, gamma exposure rates may be sufficient to prevent unlimited land use and, even if enough stabilizing materials were used to suppress the gamma radiation, radon exhalation probably could prevent unrestricted land use also. Some of the possible radiation problems could be reduced by separating the waste rock and sub-ore when hauled to the surface. The waste rock could then be used as a blanket for the sub-ore. Away from the pit proper, surface gamma readings must be below 62 μR/hr to comply with Texas State regulations. It is reasoned that, since background is about 5 μR/hr, surface gamma radiation of 57 μR/hr or less would cause a total body dose of 500 mrem/yr or less.

A number of the older mines in Texas were active in the late 1950's and early 1960's--before there were requirements for stabilization. Such

Figure 6.12 Aerial view of the Manka Mine, Karnes County, Texas. Note the extent of the mine pit and associated waste piles with poor vegetative growth on bare wastes or those with insufficient topsoil cover.

Figure 6.13 Overburden pile showing the weak vegetative cover and gullying associated with improper stabilization at the Manka Mine, Karnes County, Texas. Mine stabilized in 1974.

mines, one of which is shown in Fig. 6.14, are relatively shallow, contain shallow pools of water, and have high associated gamma radiation on the order of 80 to 100 μR/hr and as much as 140 to 250 μR/hr in some areas. The particular mine in Fig. 6.14 has maximum readings of 400 μR/hr on the mine waste piles. In addition, the mine was used for illegal disposal of toxic wastes, primarily styrene, tars, and unidentified ceramic or refraction nodules. Some of the drums containing the wastes are shown in the rear center and right of the photograph.

Mine wastes may be used for construction and other purposes if they are not controlled or restricted (see Sections 5.4 and 6.1.5.3.1). These wastes have been used for fill in a yard and park (Appendix G). Possibly they have also been used in a school area and fairgrounds (Th79). Their use in dwelling construction has also been reported (Ha74). It is also common practice to use mine wastes for road ballast and fill in areas around mine sites. This type of usage is evident from the roads immediately adjacent to and located north and northeast of the mine shown in Fig. 6.3.

In summary, only about six percent of the land used for uranium mining has been reclaimed from 1930-71 (Pa74). For the most part, the wastes at the mine sites are spreading as a result of weathering and erosion. It appears that the wastes can be controlled or disposed of by altering some mining practices, which would require very little effort or expense on the part of the mining industry. Any reclamation of the mine sites should be keyed to long-term, natural reclamation that will continue indefinitely. Careful planning can hasten the natural reclamation process and insure long-term stability of the mine sites. Measures should be taken to prevent the removal of mine wastes.

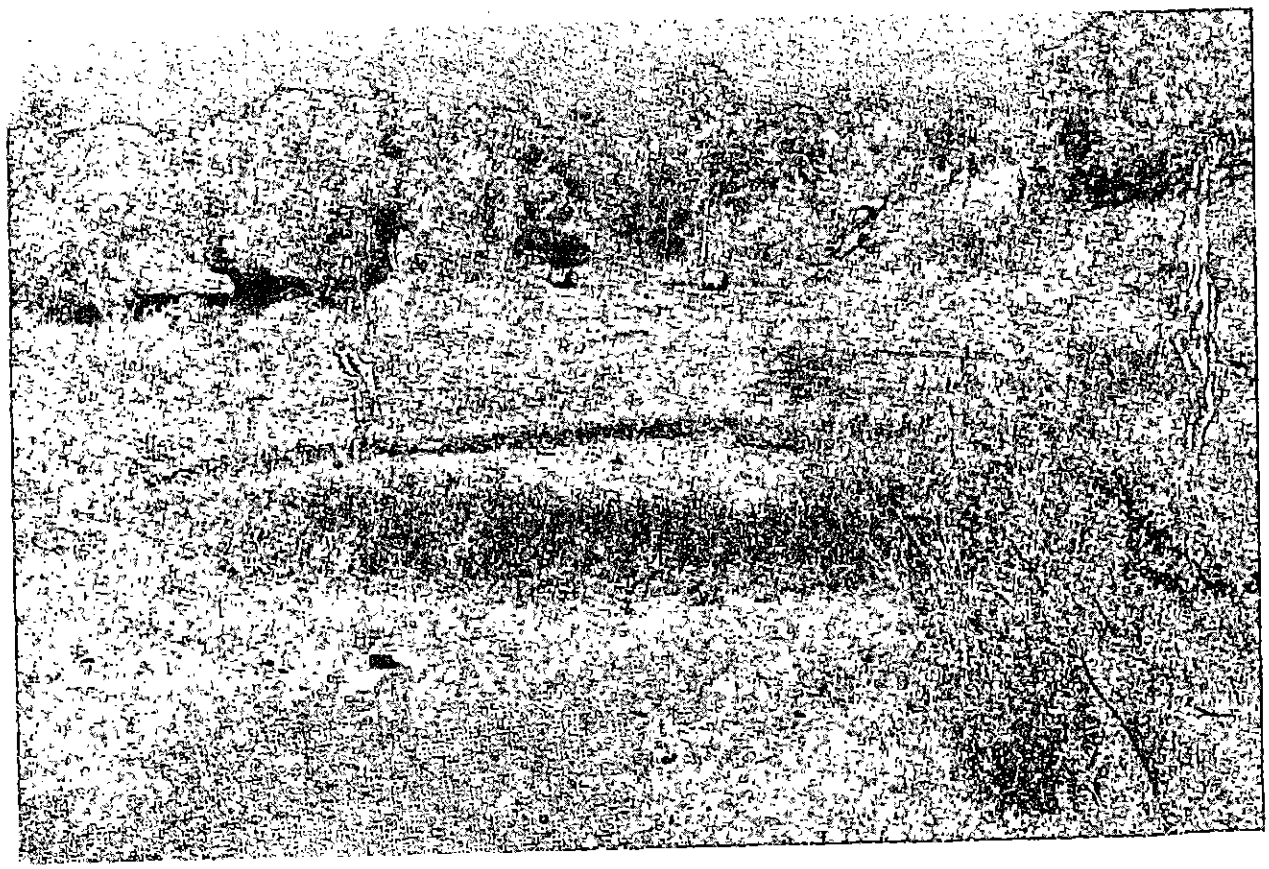

Figure 6.14 Inactive Hackney Mine, Karnes County, Texas. Drums in background contained toxic liquid wastes and styrene. Mine was active in late 1950's and early 1960.

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